SLASH ADDITIONS: A TOOL FOR RESTORING HERBACEOUS COMMUNITIES IN DEGRADED PINYON-JUNIPER WOODLANDS

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ABSTRACT

SLASH ADDITIONS: A TOOL FOR RESTORING HERBACEOUS COMMUNITIES IN DEGRADED PINYON-JUNIPER WOODLANDS

Michael T. Stoddard

Trees in pinyon-juniper woodlands are encroaching into adjacent intercanopy spaces, developing a continuous canopy structure with high fuel loads and a decline in herbaceous production and species richness. Increases in tree density can contribute to the depletion of essential soil nutrients and moisture from nearby intercanopy spaces, thereby affecting the establishment of perennial grasses and forbs within these interspaces. We established an experiment within interspaces of two pinyon (*Pinus edulis*) and juniper (*Juniperous osteosperma*) woodlands to examine the effects of slash and seed additions on soil stability, soil chemistry, soil biota, and graminoid establishment. The study site was in the Grand Canyon-Parashant National Monument, at Mt. Trumbull, Arizona on both cinder and sedimentary soil types. Our goal was to decrease sediment loss and create favorable microsites for soil biota and graminoid seedling establishment. Slash additions increased residual woody and litter debris, which decreased rates of sediment loss. Changes in soil nutrients were not observed, however available $N0_3$ decreased significantly with slash additions, suggesting increased microbial activity. Arbuscular mycorrhizal fungi and microbial carbon biomass increased significantly as a result of slash additions. Graminoid cover increased over 200% across both sites in slash and seed treatments compared to seed only treatments. In the second year, 42% of the slash and seed plots contained at least one reproductively active seeded

graminoid species. Nineteen percent of the total cover (14.3 %) was comprised of the seeded graminoid species. These results indicate that slash treatments do create favorable microsites for graminiod emergence and establishment, therefore contributing to increase herbaceous production within pinyon-juniper woodland interspaces.

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"One of the penalties of an ecological education is that one lives alone in a world of wounds. Much of the damage inflicted on land is quite invisible to laymen. An ecologist must either harden his shell and make believe that the consequences of science are none of his business, or he must be the doctor who sees the marks of death in a community that believes itself well and does not want to be told otherwise". (Aldo Leopold 1953)

PREFACE

There are four chapters in this thesis: introduction, literature review, journal manuscript and a conclusion with management recommendations. This thesis is organized in manuscript format, meaning redundancy is inherent among chapters. Plant nomenclature followed the USDA Plants Database (USDA, NRC 2004).

CHAPTER 1

INTRODUCTION

Pinyon-juniper ecosystems are often considered the region of transition between coniferous forests and desert ecosystems, occupying over 30 million hectares in western North America (West 1984, Tausch 1999). Pinyon-juniper ecosystems are highly variable in regards to their composition and structural makeup (West 1984, Brown 1994, Romme et al. 2003). The variability in overstory structure and composition greatly depends on site characteristics and historical factors such as fire and past management history (West 1999).

Well documented examples demonstrate that pinyon-juniper ecosystems are increasing in both distribution and density throughout their range, invading neighboring grasslands, sagebrush steppes and meadows (Young and Evans 1981, Jameson 1962, Allen 1989, Miller and Rose 1995). This densification of overstory structure can have negative impacts on understory abundance and richness (Tausch et al. 1981, West 1999), and lead to a decrease in availability and quality of water (Archer 1994, Wu et al. 2001). The diverse structure of pinyon-juniper ecosystems are being lost and herbaceous patches are developing into late sucessional woodlands (Evans 1988), causing increased risks of stand replacing fires (West1999, Miller & Tausch 2001). There have been many efforts in the past to reverse the trend of degradation caused by the increasing number of trees, mostly through the process of eradication (Arnold et al. 1964, Clary et al. 1974, Tausch and Tueller 1977). Recently, there has been more focus on restoring the structure, composition and function of pinyon-juniper ecosystems (Jacobs and Gatewood 1999, Brockway et al.. 2002). The restoration of an ecosystems aims at reestablishing the range

of historic variability that is characterizes an ecosystem prior to degradation (White and Walker 1997, Sauer 1998, Romme et al. 2003). Therefore, the process of assisting the recovery of an ecosystem should rely heavily on site-specific knowledge.

My study sites were located in the Uinkaret Mountains within Grand Canyon-Parashant National Monument at the base of Mount Trumbull, Arizona. The Uinkaret Mountains are a series of extinct volcanoes that extend 2438 m above sea level (Friederici 2003). This "sky island" receives on average 42.9 cm of precipitation a year (RAWS). However, extreme fluctuation in yearly precipitation occurs in this area. Precipitation patterns follow a bimodal pattern in which the majority of the precipitation falls within the winter and summer months (Brown 1994).

Native Americans inhabited the Uinkaret Mountains for thousands of years (Kelly 1934, Altschul and Fairly 1989). Hundreds of archaeological sites have been found around the Mt. Trumbull area (Altschul and Fairly 1989). The majority of these archaeological sites are within the current pinyon-juniper transition zone (R.Davis and A.Wilkerson pers. comm.). It is estimated that the number of inhabitants could have exceeded 1000 (Altschul and Fairly 1989). Their impacts on the pinyon-juniper transition zone include the establishment of agriculture fields, extensive fuel wood harvesting and the utilization of fire to manage certain species and ideal habitat conditions for hunted species (Kelly 1934, Steward 2002).

Euro-Americans settled the area in 1870 (Altschul and Fairly 1989). Shortly thereafter a small sawmill was constructed at the base of Mt. Trumbull mainly to aid in the construction of the St. George Mormon Temple (R.Davis, pers. Comm.). The Mormon Temple Trail was constructed to transport logs from Mt. Trumbull to St.

George. Remnants of this trail pass directly through one of our study sites. Intensive livestock grazing was also introduced in 1870 and continued well into the 1960s (Altschul and Fairly 1989). In 1961, U.S Forest Service reported that grass cover was depleted due to uncontrolled grazing before 1900 and had not yet recovered (unpublished report, BLM District Office, St. George, UT). A range inspection report from 1969 stated that all three allotments within the area were in poor condition and grass species were almost 100 % utilized, every year (unpublished report, BLM District Office, St. George, UT). Repeat aerial photographs between 1940 and 1992 and historical maps indicate that a water catchment and a water pipeline were built near our study sites. Intensive grazing clearly had an affect in reducing the native grass communities and impacted the soil quality. Today, herbaceous communities remain sparse with less than 7 % total herbaceous cover (Huffman et al., unpublished report). The reduction of understory species most likely influenced the expansion of pinyon and juniper species into neighboring herbaceous patches. Currently, the average number of trees per hectare (TPH) at the two study sites is 638 and 832, respectively (Huffman et al., unpublished report). It is estimated that tree densities in 1875 ranged between 104 and 261 TPH (Huffman et al., unpublished report). The current conditions suggest that these sites are in an advanced state of degradation and intensive management is needed in order to aid the recovery of understory communities.

Utilization of pinyon-juniper woody material is often limited to firewood or fencing material and, in many cases, is left unused. Tree thinning can produce large amounts of woody material that is often piled and burned on site, a practice that can cause soil sterilization and long-term ecological damage (Neary et al. 1999, Haskins et al.

2004). However, dispersing slash into the intercanopy area is a method of woody biomass utilization that has been shown to enhance site conditions for herbaceous plant establishment (Jacobs and Gatewood 1999).

The purpose of this study is to create favorable microsites that restore grass species and other understory components in the interspaces of pinyon-juniper woodlands. The specific research question for this study was: Does slash addition retain soil resources such as soil nutrients and soil biota and increase rates of graminiod seedling establishment. The objectives of this research were to: 1) establish soil surface amendments using on-site residual woody material, 2) measure key microsite abiotic and biotic variables, and 3) determine the effects of slash additions and seeding in pinyon-juniper intercanopy spaces.

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CHAPTER 2

LITERATURE REVIEW

Introduction

The distribution of pinyon-juniper ecosystem "type" covers approximately 30 million hectares in semi-arid regions across the western U.S, making it the third largest ecosystem in North America (West 1999a). Pinyon-juniper ecosystems are found within the Great Basin, Colorado Plateau, Rocky Mountains, Mohave, Sonoran and Chihuahuan Deserts (Brown 1994). These particular ecosystems are broadly defined as having one drought-tolerant juniper (*Juniperus*) and /or one drought tolerant pine (*Pinus*) subsection Cembroides and are typically found within an elevational gradient of 1370-2290 m (Gottfried et al. 1995, West 1999b). Pinyon-juniper ecosystems are the most xeric of the forest types in the United States (Budy and Meeuwig 1987). Most pinyon and juniper species are extremely slow growing and live to approximately 400 and 1000 years, respectively (Swetnam & Brown 1992, Graumlich 1993). Currently, eight species of pinyon and sixteen species of juniper species occupy pinyon-juniper ecosystem types (West 1999a). Understory composition of these vegetation types is highly variable, often composed of species from adjacent grasslands, shrub steppes, chaparral and montane forests (West et al. 1975). The extensive range and variability of these ecosystems, can provide many important ecosystem services such as watershed and hydrologic values as well as diverse wildlife habitats (Clary et al.1974, Evans 1988, Roundy and Vernon 1999).

Climatic fluctuation within the past 11, 000 years has had variable effects on the expansion of pinyon-juniper ecosystems (Betancourt et al. 1990, Miller and Wigand

1994, Swetnam et al. 1999). Ongoing plant migration characterized the Holocene and most likely has continued into the modern era. Evidence such as packrat middens and pollen fossil records indicate that woody species expansion was well underway before Europeans first entered the region (Betancourt et al. 1990, Swetnam et al. 1999). Therefore woody population expansion and densification may confound inferences about human impacts on present invasion. However during the last 150 years, pinyon and juniper populations have undergone extensive range expansion and densification. In southwestern Utah, pinyon and juniper tree densities have increased by 6 to 20 times (Cottam and Stewart 1940). In Nevada, surveys indicated that pinyon and juniper populations have increased almost 2.5-fold (Tausch et al. 1981). Miller and Wigand (1994) concluded that more juniper exist today than any time period in which records are available. The primary factors that have most likely contributed to the recent rapid expansion of theses woody species are optimal climatic conditions, the indirect effects of European settlement, (i.e. livestock grazing, elevated CO_2, and fire exclusion) and the removal of native America practices such as the use of fire and firewood harvesting (Allen 1989, West 1999a, Stewart 2002).

At the landscape level, pinyon and juniper ecosystems are composed of a series of patch and matrix assemblages where the matrix is the dominant feature and the patches are embedded in between the matrix (Tausch 1999). These patch and matrix assemblages vary with the distribution of overstory and understory vegetation structures (Tausch 1999, Romme et al. 2003). Three general pinyon juniper vegetation structures have been identified based on these overstory and understory relationships (West 1999a, Romme et

al. 2003). These structural types include: pinyon-juniper grass savanna, pinyon-juniper shrub woodland and pinyon-juniper forest.

Pinyon-Juniper Grass Savanna

Pinyon-juniper grass savannas are characterized as having a matrix dominated by perennial bunch grasses and a few pinyon and juniper trees dispersed as small patches or isolated individuals (Tausch 1999). Historically, the structure of savannas was maintained by frequent, low severity fires that thinned and killed encroaching trees (Dwyer and Piper 1967, Young and Evans 1981, Allen 1989). These fires were primarily surface fires carried by perennial grasses that burned between mid spring and the late summer (Young and Evans 1981, Allen 1989, Miller and Rose 1999). Human impacts on these savannas date back well into the early hunter and gather period (100 B.C) (West 1999a). Native Americans use of fire and harvesting of firewood may have favored savanna structures over woodland structures on the more gradual slopes (Stewart 2002). As Native American populations increased so did the intense usage of resources, especially in the Southwest (Kohler 1988). It is hypothesized that over utilization of resources and climate change may have contributed to the downfall of the Pueblo civilization (Denevan 1992b, Scurlock 1998). Spanish settlers introduced livestock grazing in the late 1500's (Allen 1989). It was not until the late 1800's, after European settlers arrived that herbaceous species became depleted due to uncontrollable grazing (Taush 1999). Intense grazing pressures may reduce the fine fuels, and allow trees to multiply and invade into these savannas (West 1984, Young and Evans 1981, Allen 1989). Tree species have extended into these savannas, potentially altering the structure and key functional processes that maintain these open vegetation structures (Tausch et al.

1981, West 1984, Miller and Rose 1995). This process has resulted in a discontinuous fine fuel layer and thereby decreased the fire frequency, which governs the expansion of woody species (Allen 1989, Miller and Wigand 1999, Savage and Swetnam 1990). Climatic changes have also apparently influenced the expansion of woody species (Betancourt et al. 1991, Miller and Wigand 1994). As a result, some of today's pinyon-juniper savannas are no longer savannas and are characterized by a dense late successional woody canopy structure with little herbaceous cover. More intense fire behaviors are associated within this vegetation type thereby increasing the risk of invasion by exotic annual herbaceous species (West 1999).

Pinyon-Juniper Woodland

Pinyon-juniper woodlands are distinguished by a heterogeneous structure where trees are the dominant vegetation and patches of understory vegetation are distributed between the tree matrix (West 1984, Romme et al. 2003). Overstory effects on shading, rain interception, nutrient concentration and moisture uptake create a range of soil microsites for understory species establishment (Breshears et al. 1998). This mosaic structure can produce an array of favorable conditions for a variety of species. For example, several cool season grasses are more prevalent underneath or adjacent to tree crowns of alligator juniper (*Juniperous deppena*) (Clary and Morrison 1973). Pinyon pine establishment is often associated with a nurse plant that provides optimal conditions (Stulz 2004). Some perennial species seem to favor the transition zone between the intercanopy space and underneath tree canopies (Everett and Koniak 1981). As the overstory structure becomes more dense, these woodlands become more homogenous, losing the variety of soil microsites associated with pinyon juniper woodlands.

Pinyon-juniper woodlands are increasing in tree density probably due to many of the same environmental and anthropogenic impacts that have been altering savanna structures. Climate shift of both wet periods and extended droughts have contributed to increasing tree densities (Norwak et al. 1994, Miller and Wigand 1994). Native Americans have impacted these lands for thousands of years, utilizing both understory and overstory species (Alcoze and Hurteau 2001). Intensive grazing pressures introduced in the mid 1800s, have reduced the abundance and richness of many understory species (Taush and West 1995, Tausch et al. 1981). This reduction of understory species has likely allowed woody species to encroach into intercanopy spaces (Miller and Wigand 1994). Increases in woody cover have contributed significantly to the depletion of soil moisture and available nutrients within the soil (Breshears 1997a, Schlesinger et al. 1990). With the removal of understory species, sufficient ground cover to stabilize soil resources has been reduced. These exposed soils are then susceptible to sheet erosion, reduced infiltration, and the inability to form soil aggregates (Wood et al. 1987). In addition, soil seed banks are depleted with the loss of surface soils, thereby compounding the degradation problem in pinyon-juniper ecosystems (Jacobs and Gatewood 1999). Woodlands have become more homogenous as tree species establish within the intercanopy patches (West et al. 1979). Rapid overstory densification has had adverse effects on the understory abundance and species richness within these woodland ecosystems (West 1993).

The difference between pinyon-juniper woodlands and savannas are ultimately based on the ratio of overstory and understory structures. The distinctions between these two structural types are subjective and often are dependent on management objectives.

The role of fire also seems to differ between these two structural types (West 1999a Romme et al. 2003). Historical fires in pinyon-juniper woodlands were often less frequent and more intense than fires in savannas (West 1999, Romme et al. 2003). Fires in woodlands can be potentially classified as either a mixed-severity fire regime (Brown 2000), where patches of low, moderate and high intense fires create a mosaic of fire effects or a high-severity fire regime (Baker and Shinneman 2003). These effects and severities are highly dependent on climatic and stand conditions. It has also been documented that Native Americans actively used low intensity fires to maintain large herbaceous patches within woodlands in order to provide a reliable food source (West 1999, Stewart 2002). Fire histories and the effects of fire in pinyon-juniper woodlands are poorly understood (West 1999, Baker and Shinneman 2003). What is apparent, larger and more intense fire behaviors are associated in today's pinyon-juniper woodlands compared to pre Euro-American settlement (West 1999). Because soil seedbanks and disseminating native perennial plants have been depleted, (Taush and West 1995, West 1999) opportunistic exotic species are invading after intense disturbances, changing the structural composition and diversity of these woodlands.

Pinyon-Juniper Forest

Pinyon-juniper forests are described as having an almost closed tree canopy structure with little to no understory component (West 1999a, Romme et al. 2003). Old growth characteristics, which include dense, multistory canopies with old-aged trees, typify pinyon-juniper forests (Miller et al. 1999). The patch and matrix assemblage is composed of a very dense overstory structure separated by bare ground or sparse understory patches similar in size to the overstory matrix (Romme et al. 2003). It is

estimated that 3 to 5 % of pinyon-juniper ecosystems are old-growth forest types (Miller et al. 1999). These forest types are typically found in areas of rugged and isolated topographical features (Waichler et al. 2001, Miller and Rose 1999). Individual old-growth trees are characterized as having irregular canopies with large diameter trunks (Burkhardt and Tisdale 1969, Miller et al. 1999). Pinyon juniper forest types can support fuel loads that are patchily distributed (Romme et al. 2003). Discontinuous fuel compositions, where heavy fuels are distributed within discrete patches do not carry fire readily. Low abundance of fine fuels such as grasses and forbs greatly reduce the rate of fire spread. Fire regimes within these forest types are considered to be infrequent and stand replacing (Floyd et al. 2000, Floyd et al. 2004, Baker and Shinneman 2004). The reason for an infrequent fire regime is not due to the lack of fire ignitions. Lightning ignites fuels within pinyon-juniper forest types as often as any other pinyon- juniper vegetation types, but the lack of continuous fuels and the isolated topography prevents fires from spreading (Omi and Emrick 1980, Romme et al. 2003, Floyd et al. 2004). The vast majority of fires within pinyon-juniper forest types burn no more than an individual tree or isolated patches of trees (Omi and Emrick 1980, Romme et al. 2003). Fires in these forest types can spread from one overstory patch to the next only under weather conditions that exhibit strong winds and low fuel moisture (Floyd et al. 2004, Romme et al. 2003). Under extreme weather conditions, fire spreads through the crowns of overstory species, generating severe fire behavior with rapid rates of spread (Floyd et al. 2004). It is theorized that these particular forests have undergone few structural changes due to fire exclusion in the last century and probably are not outside the historical range

of variability in terms of stand structure, fire frequency and fire behavior (Floyd et al. 2004, Romme et al. 2003).

Management in these pinyon-juniper ecosystems often strives to gain a more productive understory by eliminating overstory trees (McArther and Young 1999). This can be considered the conversion of one vegetation type to another. Management techniques such as chaining, pushing, burning and application of herbicides are utilized to increase perennial grasses, specifically for livestock forage (Campbell 1999). Pinyon-juniper ecosystems are highly variable in many aspects yet are often managed for similar goals and objectives. This simplification of management has contributed to the loss of biological diversity that is associated within pinyon-juniper ecosystems.

Ecological Restoration of Pinyon-Juniper Ecosystems

Densely populated pinyon-juniper stands occupy tens of thousands of hectares across the western United States (Taush et al. 1981, West 1999a). Tree density has increased rapidly in a short time period in both pinyon-juniper savannas and woodlands (Romme et al. 2003). The advance of tree species into adjacent intercanopy spaces has had negative impacts on the understory composition and structure (Miller and Wigand 1994, Taush and West 1995). Soils have become unstable due to the lack of ground cover and soil infiltration rates are below historic levels adding to the depletion of base flows into intermittent streams (Archer 1994, Thurow and Carlson 1994, Wu et al. 2001). Allen (1989) proposed that different trajectories of succession may not allow degraded ecosystems to return to a form of its original state. The reasons may be attributed to a lack of seed or seed sources, climatic change, dominance by a highly competitive species, or accelerated erosion (Allen 1989). The causes for these altered states are often blamed

on anthropogenic factors such as grazing and fire suppression, which potentially has led to the densification of tree species. Friedel (1991) points out that once a system crosses a threshold to a more degraded state, improvement cannot be attained by simply removing the causes of degradation. Management intervention is needed to transition from one state to another (Friedel 1991). Therefore without management, this episode of accelerated soil erosion and overstory densification appears to be highly persistent and irreversible (Davenport et al. 1998). With pinyon-juniper ecosystems in an advanced state of degradation, ecological restoration has recently become an explored management option for these ecosystems (Jacobs & Gatewood 1999, Brockway et al. 2002, Huffman et al., unpublished data, Landis and Bailey 2005).

The term "ecological restoration" has often been loosely applied to management within pinyon-juniper settings. Applications that improve understory communities and remove pinyon and juniper trees (i.e., "chaining") to enhance range conditions for livestock can often be mistaken as a form of restoration application. Removal of pinyon and juniper trees to create grasslands is not grassland restoration but rather the conversion of one ecosystem type to another. Thinning and burning applications to reduce overstory and fire hazards can also be misguided restoration efforts (Baker and Shinneman 2004, Romme et al. 2003). These prescriptions often lack clear site-specific objectives based on reference conditions (Landis and Bailey 2005).

The Society of Ecological Restoration (2004) defines ecological restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed". This definition implies that species comprising an ecosystem are adapted to function within a specific range of variability over periods of evolutionary time

(Stephenson 1999). Degradation of an ecosystem happens when attributes are altered, often due to human activities, and take on characteristics outside long-term ranges of historic variability (Fulé et al. 1997, Sauer 1998). The recovery of ecological attributes or functions reestablishes the range of historic variability that is unique to an ecosystem prior to degradation. Good ecological restoration should not only entail the attempt to reestablish specific ecosystem characteristics but should also address diverse social perspectives (Higgs 1997). Therefore the process of assisting the recovery of an ecosystem based on ecological knowledge is just as important as the products, which are the diverse perspectives of the interested stakeholders (Higgs 1997). This allows human values to be part of the restoration equation.

A vital part of restoration principles is the use of reference conditions to steer management objectives (Fulé et al. 1997, Moore et al. 1999). Reference conditions represent a spectrum of ecosystem conditions, where the ecosystem structure, composition and key functional processes behave within a range of historical variability (Laycock 1991). The concept of historical range of variability addresses the dynamics of ecosystems undergoing patterns of change (Morgan et al. 1994). Reference conditions can be used to help formulate treatments and create benchmarks to evaluate the success of restoration treatments (Fulé et al. 1997, White and Walker 1997). These treatments or targeted benchmarks should be adaptive in order to address ecosystems dynamics.

Several limitations exist when trying to restore sites based on reference conditions. First, the historic range of variability in a particular ecosystem can be difficult to assess especially in regards to disturbance events (Fulé et al. 2002). Second, reference conditions can be limited to a few elements of an ecosystem and the

complexities across a landscape can vary greatly in terms of soil type, community structure, composition, nutrient cycling and functional processes that maintain the ecosystem such as fire or hydrological processes (Fulé et al. 1997, Landres et al. 1999). Tausch (1996) points out a third limitation that past climatic changes may have influenced the increases in woody vegetation, therefore limiting the applicability of past reference conditions to present day ecosystem conditions (Millar and Wolfenden 1999). Westoby et al. (1989) discuss a fourth limitation, ecosystems undergo multiple stable states. Transitions between one state to alternative states can be triggered by natural events (i.e. weather, fire) or by management actions (i.e. tree removal) making it difficult to decide which stage (pre or post disturbance) to benchmark. Ecosystems undergo multiple states and time is an essential component of ecological restoration principles. Therefore historical reference conditions are not meant to be a blue print, but are intended to provide a foundation to assist in the recovery of an ecosystem (Aronson et al. 1993). The use of reference conditions in ecological restoration applications is not meant to reverse time, rather to help identify ecological characteristics of healthier and more intact ecosystems (Moore et al. 1999). These conditions are then used to guide management and treatments. Time periods that create favorable states can be labeled as opportunities. Restoration aims at seizing these opportunities. Confidence in reference conditions can be improved when using multiple lines of evidence to address historical patterns (White & Walker 1997, Swetnam et al., 1999). Sources such as historical records, on-site physical evidence, ecological legacies, relict information and traditional indigenous knowledge can provide valuable information to establish reference conditions (Fulé et al. 1997, Sauer 1998 Alcoze and Hurteau 2001).

Legacy studies document past geographic distribution of species, communities and ecosystems. This could be helpful in disentangling natural versus anthropogenic changes (Rundall et al., unpublished report). Floyd et al. (2004) emphasizes that stand replacing fires were a natural function of pinyon-juniper forests at Mesa Verde National Monument and these forests are not outside the range of natural variability in terms of stand structure, composition and functional processes. Allen and Breshears (1998) described the rapid increase in pinyon-juniper woodland distribution and densification throughout New Mexico. Legacy studies are important resources that have increased our understanding of successional cycles in nature, and provided reliable information for management decisions (Swetnam and Betancourt 1998).

Relict sites are areas that have undergone little to no change caused by anthropogenic disturbances (Johnson 1962, Thatcher and Hart 1974). Several pinyon-juniper relict sites have been surveyed including, Boysag Point, Fishtail mesa, No Mans mesa and Spy mesa. These are isolated mesas found within and surrounding Grand Canyon National Park, Arizona (Jameson et al. 1962, Schmutz et al. 1967, Thatcher and Hart 1974, Rowlands and Brian 2001). The primary interest of these relict site studies was to collect base line information on the structure and composition of the vegetation community, soil characteristics and fire evidence within intact or undisturbed communities (Jameson et al.1962, Rowlands and Brian 2001).

Three studies have been implemented to improve pinyon-juniper ecosystem conditions based on ecological restoration principles. Reference conditions were assessed and developed differently for each study. First, Jacobs and Gatewood (2002) initiated a project to evaluate the efficacy of restoration techniques for reestablishment of

native herbaceous cover at a site in north-central New Mexico. Specifically, they wanted to determine if overstory reduction and slash mulching treatment would yield a positive herbaceous response. Historical evidence suggested that these areas were intensely grazed beginning in the late 1880s denuding many of the herbaceous communities. Tree thinning treatments were based on age class information from a nearby pinyon-juniper savanna. Evidence suggests an exponential increase in young pinyon and juniper stem densities, and spatial evidence of older growth trees. Overstory tree removal prescriptions were carried out to maintain a 15-20m spacing between mature trees, which was considered optimal for restoring pinyon-juniper savanna characteristics. Results after three years showed significant increases in total herbaceous cover and a reduction of soil erosion rates (Jacobs and Gatewood 1999). Brockway et al. (2002) also initiated a project to evaluate the effectiveness of mechanical overstory reduction and three slash treatment alternatives followed by prescribed fire as techniques for restoring grassland savannas from degraded woodlands. They chose to reduce juniper tree densities to 15 per hectare. After two years, understory percent cover was significantly greater in treated plots compared to control plots. There were no significant differences in the alternative slash treatments suggesting that understory increases were primarily the result of the decrease from overstory competition. In an unpublished report, Huffman et al. implemented a study to test ecological restoration principles applied to pinyon-juniper ecosystem on the Arizona strip near my study sites. Objectives were to decrease overstory structures to levels similar to reference conditions and increase understory productivity and diversity. Small trees (below 25 cm diameter at root collar DRC) were cut, thinned slash material was dispersed into interspaces and native plants were seeded.

Treatments appeared to be effective in reestablishing overstory composition and density similar to reference conditions. Slash dispersal, seeding treatments and the reduction in tree density did not significantly increase understory abundance and richness compared to control units one year following the application of all treatments.

These three studies had similar goals, which were to reduce overstory densities and increase understory production based on the best information available. All three projects clearly reduced overstory densities. Two out of three increased herbaceous understory communities. Only one project (unpublished report, Huffman et al. 2005) had clearly stated reference conditions in which restoration treatments were formulated from and then targeted. The other two projects had predetermined structures that they were trying to mimic. According to Landis and Bailey (2005), age structure and spatial arrangement or clumping of pinyon and juniper trees should be considered when trying to restore particular overstory structural types. Uniform prescriptions for tree reduction in pinyon-juniper ecosystems could reduce the variability associated with these ecosystems. Assess restoration efforts is somewhat subjective, but as demonstrated within the following examples multiple lines of evidence for tree reconstruction should be considered before applying restoration prescriptions.

Resource Island Hypothesis and Legacy Effects

Water and soil nutrients are in short supply in arid and semi-arid ecosystems (Noy-Meir 1973). These resources vary both temporally and spatially in distribution (Noy-Meir 1985). The spatial distribution of these resources plays an important role in the establishment and distribution of vegetation in these harsh environments (Schlesinger et al. 1994). Arid ecosystems can have a high diversity of perennial plants that usually

occur in patches (Schlesinger et al. 1994). These patches can promote a diverse community of microbes, soil microfauna, and mesofauna, which are key factors in water infiltration rates and nutrient cycling (Gallardo and Schlesinger 1995, Herman et al. 1995). The availability of essential soil nutrients is in large part controlled by the below ground biota, which regulates the mineralization and immobilization of these nutrients (Schlesinger et al. 1994). These processes determine the availability of important nutrient uptake thereby regulating vegetation growth and establishment.

Many southwestern arid and semi-arid ecosystems in the United States are undergoing a desertification process that is reducing the diversity of plant communities (Herman et al. 1995). The desertification of southwestern ecosystems is usually associated with the conversion of grasslands to a woodier ecosystem or the loss of herbaceous patches within woodlands. Associated with this conversion is the redistribution of soil resources and water availability from a relatively homogeneous pattern to a highly variable distribution that is concentrated underneath the canopies of woody species (Klopatek and Klopatek 1986, Reynolds et al. 1999). As grasses are replaced with woody vegetation, the cycling of soil nutrients is often confined to the zone of litter accumulation beneath the woody vegetation, while the surrounding interspaces can become depleted of soil nutrients and water (Breshears et al. 1994, Pieper 1990). Trees and shrubs can extract nutrients and water from adjacent intercanopy soils, and then redeposit those resources underneath their canopies (Doescher et al. 1987, Breshears et al. 1997a,), causing bare interspaces soils to become more susceptible to the loss or reduction of topsoils (Wilcox and Breshears 1995, Davenport et al. 1998). Seed resources are then redistributed from the interspaces and trapped within the organic

accumulation underneath the woody vegetation (Wilcox and Davenport 1995, Chambers 2000).

Several conceptual models address these processes as the formation of "resource islands" or "islands of fertility" (Charley and West 1975, Schlesinger et al. 1994, Reynolds et al. 1999). These islands are preferred sites for the regeneration of woody species, herbaceous plants and various soil biota that promote nutrient cycling (Lugwig et al. 1988, Herman et al. 1995). These islands can serve as obstructions where organic matter is trapped and made available for mineralization (Herman et al. 1995). These processes produce a positive feedback loop that reinforces the cycling of resources within and around the island, allowing the island to expand and promote the establishment of other perennial plants (Schlesinger et al. 1990). Herman et al. (1995) concluded that soil microorganism populations will be higher where soil resources are higher. Therefore in shrublands where soil recourses have a patchy distribution, soil biota seems to concentrate underneath or around woody vegetation where soil resources are higher. Tongway and Ludwig (1996a) tested the resource islands hypothesis by creating artificial islands with the use of dead *Acacia aneura* branches. As a rehabilitation treatment on bare soils, branch piles had significant positive effects on soil erosion, water infiltration, soil respiration and accumulation of soil nutrients. Soil erosion was reduced considerably in branched plots compared to non-branched plots. Water infiltration and soil respiration were significantly higher in branch-treated plots. The increase in soil respiration indicates an increase in CO_2 release, which indirectly indicates an increase in microbial activity (Sclesinger et al. 1994). This increase in soil activity is most likely an effect of woody materials stimulating a wide range of biological interactions (Tongway and Luwig

1996a). Tongway and Luwig (1996b) concluded that patches constructed of tree branches can function as refugia for vegetation during droughts.

Microsite and Seeding Dynamics

Perennial grass densities have decreased across the range of many arid and semi arid ecosystems, thereby increasing soil erosion and the loss of favorable microsties necessary for germination and establishment (West 1984, Chambers 2000). Natural revegetation of perennial species is often slow or unsuccessful due to the loss of soil seedbanks and reproductively viable vegetation to produce seeds (Poulsen et al. 1999, Stevens 1999). Seeds are often sown in combination with various soil treatments or sown directly on the soil surface (Fowler 1988, Call and Roundy 1991, Chong 1994). Surface sown seeds are exposed to many environmental hazards that often result in limited germination success (Monsen and Stevens 1999, Chambers 2000). Due to escalating seed costs and the depletion of soil resources, more effective means of creating ideal microsite conditions are needed to stimulate seed germination and seedling establishment (Monsen and Stevens 1999, Poulsen et al. 1999)

Seedling germination and establishment is directly correlated with the number of seeds in favorable microsites, rather than the total number of available seeds (Harper 1977, Harper et al. 1965). One of the most important elements that determine successful microsites for seed germination and establishment is the association with high humidity and high moisture (Harper et al. 1965). Also understanding the movement and fate of seeds once the seed has been dispersed is essential when trying to restore disturbed ecosystems (Chambers 2000). Significant movement of natural and man-dispersed seeds can occur before a seed finds its final resting site (Chambers 2000). Wind is a prominent

feature within many arid and semi arid ecosystems and with less ground cover, seed movement along the surface becomes greater. Eventually seeds can become trapped in areas that create some sort of aerodynamic drag such as, vegetation, litter and duff layers and dead and down woody material (Fowler 1986, Stamp 1989b).

In conclusion, land managers and researchers are currently focusing on ecological restoration as a management tool to help address problems in pinyon-juniper ecosystems (Jacobs and Gatewood 1999, Brockway et al. 2002, Landis and Bailey 2005). In this study, I was specifically interested in restoring native perennial patches associated within mature pinyon-juniper woodlands. Understory species play an important role in contributing to the function and stability of an ecosystem by: 1) providing biodiversity to an ecosystem, 2) assisting rapid nutrient turnover, and 3) increasing soil stability (West 1999b). Pinyon-juniper ecosystems are dependent upon the capacity of soil microsites to support associated plant communities. In order to successfully restore native ecosystem diversity, effective soil microsites that influence plant production need to be identified. I attempted to create favorable microsites and stimulate mechanisms that lead to plant establishment.

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CHAPTER 3

SLASH ADDITIONS RETAIN SOIL RESOURCES AND INCREASE RATES OF GRAMINOID SEEDLING ESTABLISHMENT IN A PINYON JUNIPER WOODLAND.

ABSTRACT

Trees in pinyon-juniper woodlands are encroaching into adjacent intercanopy spaces, developing a continuous canopy structure with high fuel loads and a decline in herbaceous production and species richness. An increase in tree density has contributed to the depletion of essential soil nutrients and moisture from nearby intercanopy spaces, thereby affecting the establishment of perennial grasses and forbs within these interspaces. We established a 2 X 2 full factorial experiment with two levels of seeding and two levels of slash additions within the interspaces of pinyon (*Pinus edulis*) and juniper (*Juniperous osteosperma*) tree canopies to examine the effects of slash additions on soil stability, soil chemistry, soil biota and graminoid establishment. The study site was in the Grand Canyon-Parashant National Monument, at Mt. Trumbull, Arizona and sites represented both cinder and sedimentary soil types. Our goal was to create favorable microsites or "islands" of elevated soil fertility for graminoid seed establishment. Slash additions increased residual woody and litter debris, which seem to affected rates of soil loss. Changes in soil nutrients were not observes, however available $N0_3$ decreased significantly as a result of slash treatments, suggesting increased microbial activity. Arbuscular mycorrhizal fungi and microbial carbon biomass increased significantly in slash treatments compared to non-slash treatments. Graminoid cover

increased over 200% with slash and seed additions compared to seed-only treatments. In the second year, 42% of the slash and seed plots contained at least one reproductively active seeded graminoid species. Nineteen percent of the total cover (14.3 %) was comprised of the seeded graminoid species. These results suggest that slash treatments do create favorable microsites for graminoid emergence and establishment, therefore contributing to an increase in understory production within pinyon-juniper interspaces.

INTRODUCTION

Pinyon-juniper savannas, woodlands and forests, cover 24 million ha throughout the western United States (West 1984). Approximately 40% or 9.6 million ha of the total pinyon-juniper lands occur within Arizona and New Mexico (Powell et al. 1994). Pinyon-juniper woodlands are distinguished by a heterogeneous structure where the trees are the dominant vegetation and embedded in the intercanopy spaces are species that make up the understory (Taush 1999, Romme et al. 2003). Trees in many pinyon-juniper woodlands are encroaching at an unprecedented rate into adjacent intercanopy spaces, developing a continuous canopy structure with high fuel loads and a decline in herbaceous production and species richness (Cottam and Stewart 1940, Tausch et al. 1981). The diverse structure of pinyon-juniper woodlands is being lost and herbaceous patches are developing into late sucessional woodlands (Evan 1988). Increases in the ratio of tree biomass (primarily juniper tree biomass) to herbaceous biomass is considered to be caused by climatic fluctuations favoring the establishment of pinyon and juniper trees, the exclusion of fires that removed young fire-sensitive trees, and livestock grazing (West 1999, Tausch 1999, Jacob and Gatewood 1999).

It is difficult to assess how each factor has contributed to the change in pinyon-juniper woodland structures. Evidence such as packrat middens and pollen fossil records indicate that woody species expansion was well underway before Europeans first entered the region (Betancourt et al. 1990, Swetnam et al. 1999). Therefore woody population expansion and densification may confound inferences about human impacts on present invasion. (Wigand 1987, Miller and 1994). Fire and the effects of fire are poorly understood, therefore providing little reliable evidence that low-severity surface fires maintained woodland structures (Baker and Shinneman 2004). Close temporal association between juniper tree expansion and unsustainable livestock grazing, both beginning in the late 1800s seems to suggest that livestock grazing was a major contributor of juniper expansion (Allen 1989, Gottfried et al. 1995, Taush 1999). Intense grazing pressures reduced the abundance and richness of many herbaceous communities allowing trees to establish into neighboring intercanopy spaces (Tausch et al. 1981, Allen 1989, Taush and West 1995). Increases in tree density contribute significantly to the depletion of organic carbon, nitrogen and soil moisture from nearby intercanopy spaces, therefore affecting the regeneration of perennial grasses and forbs within these interspaces (Klopatek and Klopatek 1986, Davenport et al. 1996, Breshears et al .1998). With the removal of herbaceous species, sufficient ground cover to minimize sediment runoff is also reduced (Wilcox 1994, Davenport et al. 1998, Wilcox et al. 2003a). Exposed soils are susceptible to sheet erosion, reduced infiltration, and the inability to form soil aggregates (Wilcox et al. 2003b). Wood et al. (1987) determined total herbaceous ground cover to be the single most important variable in influencing sediment runoff. In addition, the number of seeds in the soil seedbank has decreases with the loss

of surface soils, absence of seed dissemination and the increase in tree cover and density (Jacobs and Gatewood 1999, Poulsen et al 1999, Stevens 1999).

Several studies have shown that, as juniper trees are cut, herbaceous vegetation increases significantly (Aro 1971, Clary and Jameson 1981, Schott and Pieper 1985). A common solution to the changes in pinyon-juniper tree density has been wholesale tree removal across the landscape, converting woodlands into grasslands (Tausch and Tueller 1977, West 1984). This can cause further disruption to the ecosystem by eliminating trees, which provide heterogeneity of microhabitat for plant species, cover for wildlife and soil nutrient sinks. More recently emphasis of managing pinyon-juniper woodlands has shifted from forage production to improving ecosystem characteristics.

Tree densities are being reduced to levels thought to be consistent with pre Euro-American settlement (Jacob and Gatewood 2002, Brockway et al. 2002, Huffman et al., unpublished report). Utilization of pinyon and juniper woody material (slash) has shown to abate soil loss (Jacobs and Gatewood 1999, Brockway et al. 2002, Hasting et al. 2003). Brockway et al. (2002) concluded that scattering slash across harvested sites could promote herbaceous growth by fostering microsites that stabilize the soil surface. Jacobs and Gatewood (1999) found a seven-fold increase in herbaceous cover two years following overstory reduction and slash mulching treatments compared to a control treatment. Seeding was not necessary to achieve these increases in herbaceous cover. Jacobs and Gatewood (1999) found no significant increases in total grass cover resulting from the combination of seeding, tree reduction and slash mulch treatments.

Plant community recovery is possible if sufficient native understory species pools exist, both above ground and in the seedbank. Unfortunately in many pinyon-juniper

woodlands, soil seedbanks appear to be depleted, and dissemination of new seeds seems slow due to low herbaceous productivity and abundance (Taush and West 1995, Jacobs and Gatewood 1999, West 1999). Unless these woodlands are artificially seeded, natural recovery may be slow resulting in an increased time span until desired future conditions are reached (Call and Roundy 1991, Poulsen et al. 1999, Stevens 1999). However, seeding alone could be ineffective, and would result in low seedling establishment (Chong 1994, Monsen and Stevens 1999, Chambers 2000)

We hypothesized that in order for native herbaceous species (primarily grasses) to recover in many pinyon-juniper woodland interspaces, native seeds would need to be artificially introduced. Graminoid species are of particular interest because root biomass associated with grasses is likely to help in soil stabilization (West 1999). Microsites that can provide wind barriers, aid in trapping and retaining seeds, and improve soil water potential will most likely increase seed germination and establishment (Harper et al. 1965, Chambers 2000). Seed germination and establishment are directly correlated with the number of seeds in favorable microsite seedbeds, rather than the total number of available seeds (Harper et al. 1965, Harper 1977). Creating suitable microsites or "islands" of elevated soil fertility for herbaceous species, may contribute to the recovery of perennial grasses within pinyon-juniper woodland interspaces. These islands can serve as obstructions where organic matter is trapped and made available for a diverse community of microbial populations, including arbuscular mycorrhizae (AM) fungi, which are key components in water infiltration rates, development of soil structure and nutrient cycling (Allen 1991, Gallardo and Schlesinger 1995, Herman et al. 1995). Herman et al. (1995) concluded that soil microbial populations would be higher where

soil resources are higher. Microbial activity determines the availability of important nutrients thereby regulating vegetation growth and establishment (Perry et al. 1987, Gallardo and Schlesinger 1995). These processes can produce a positive feedback loop that reinforces soil resources within and around the island, allowing the island to expand and promote the establishment of other perennial plants (Schlesinger et al. 1990).

The specific objectives of this study were to determine if residual woody debris (slash material < 7.62 cm) could improve microsite conditions and increase the success of native graminoid establishment. We hypothesized that slash additions would decrease soil loss, increase soil organic levels and increase soil microbial biomass. These conditions could create islands of fertility and potentially promote the recovery of perennial graminoid within pinyon-juniper woodland interspaces.

We were also interested in evaluating an exploratory multivariate model that simultaneously examines the multiple drivers of seedling emergence in the context of a causal path model. Nutrients such as total C and N have been observed to stimulate seedling emergence of many species (Evans and Belnap 1999, Gallardo and Schlesinger 1995, Herman et al. 1995). Water availability is an essential factor in the regulation of seedling emergence (Harper et al. 1965, Harper 1977). In addition, soil characteristics such as particle size have been shown to affect seedling emergence and survivorship by influencing soil water potential (Harper et al. 1965, Chambers 2000). Slash additions has been identified as a treatment that can increase herbaceous establishment (Chong 1994, Monsen and Stevens 1999, Jacobs and Gatewood 1999). What is not exactly clear are the mechanistic processes or the relationships between slash additions and soil particle size on nutrient and water availability, consequently affecting seedling emergence. Our goal

was to draw casual relationships between these factors in order to understand the potential mechanisms influencing seedling emergence.

METHODS

Study Site

This research was conducted at two sites located within the Grand Canyon-Parashant National Monument near Mount Trumbull, AZ, U.S.A as part of a study on the effects of ecological restoration of a pinyon-juniper ecosystem. We selected two sites that represented two common soil types within this region. The first site will be referred as the clay-loam site (36N 26' 01", 113W 09' 40"). Soils at this site are within the Showlow complex, which consists of fine, smectitic, mesic aridic Argiustolls (Natural Resources Conservation Service 1982). Major management factors for the clay-loam site include slow water permeability, high shrink-swell soils, and high available water capacity (Natural Resources Conservation Service 1982). The second site, referred to as the cinder site (36N 24' 46", 113W 12' 15") is within the Wutoma-Lozinta complex which consists of ashy-skeletal over fragmental or cindery, mixed, mesic Vitrandic Haplustepts (Natural Resources Conservation Service 1982). Major management factors for the cinder site include moderate to high water permeability, and very low available water capacity. Soil particle composition is also different between sites (Figure 3.2). Soil particles were categorized into three size groups, less than 2 mm, between 2 mm and 40 mm (cobble), and greater than 40 mm. Soils were relatively finer in particle size at the clay-loam site compared to the cinder site. The cinder soils were primarily composed of cobble size soil particles. Our study sites ranged in elevation from approximately 1900 to 1950 m. Precipitation patterns follow a bimodal distribution, including monsoon

rains in July and August with snowfall in the winter months followed by a distinct dry period in May and June. The form of precipitation during June through August is primarily short-duration, high intensity storms that are associated with high rainfall erosivity. All climate data was compiled from the Nixon Flats Remote Automated Weather Station (RAWS) site at Mt. Trumbull, AZ (1980m) approximately 5 km from study sites. Precipitation was summarized by post treatment sampling years (Figure 3.3). Average daily temperatures range from -5.1 to 31.0˚ C. Vegetation at the sites is classified as Great Basin Cold Temperate Woodlands (Brown 1994). Mixed-aged pinyon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) dominate the two sites ranging in density from 638 to 832 trees per hectares. Herbaceous communities were sparse (cover < 7%) but common species consist of perennial grasses: *Bouteloua curtipendula*, *Bouteloua gracilis*, *Aristida purpurea*; perennial forbs: *Chamaesyce albomarginata*, *Eriogonum corymbosum* and *Psoralidium tenuiflorum*; shrubs: *Purshia mexicana*.

Measurements and historical evidence suggest the two sites are in a state of ecological degradation. Current herbaceous communities are sparse with less than 7 % cover and little soil O horizons are evident in intercanopy openings (Huffman et al., unpublished report). Tree densities are four times greater than a dendroecological reconstruction of the forest structure circa 1870, when Euro-American began to affect the landscape with existing tree per hectares averaging between 638 and 832. Historical evidence suggests intensive livestock grazing introduced in 1870 and continuing well into the 1960s may explain these current conditions (Altschul and Fairly 1989). U.S Forest Service reports (1961) indicate grass cover was depleted due to uncontrolled grazing before 1900 and has not yet recovered (unpublished report, BLM District Office, St.

George, UT). A range inspection report from 1969 stated that all three allotments within the area were in poor conditions and grass species were almost 100 % utilized each year (unpublished report, BLM District Office, St. George, UT)

Experimental Design

We studied unshaded, canopy openings between pinyon and juniper trees (interspaces). We implemented a 2 x 2 full factorial design with two levels of seeding (no seed, seed) and two levels of slash additions (no slash, slash). Two sites were selected and 15 interspaces were chosen at each site. At each interspace, we established four 1-m^2 (1 m x 1 m) permanent plots. Plots within each interspace were spaced at least 2 m apart to minimize between plot interactions. Plots were also located at least 3 m from the edge of a tree crown to eliminate influences created by the accumulation of organic material underneath tree canopies. We randomly assigned one of four treatments to each plot: (1) control; (2) seed only; (3) slash only; (4) slash and seed. Treatments were established in early August 2003 to coincide with monsoonal rains. Seeded treatments consisted of a mixture of four native grass species (Table 3.1). We selected native graminoid seeds based on local occurrence, baseline data from previous local studies and herbaceous community data reports from nearby relict sites. Seeds were purchased from the nearest possible seed supplier, Arizona Native Plant and Seed in Flagstaff, Arizona. Seeds were thoroughly mixed and hand broadcast before slash additions. The seed mixture was applied at a rate of 9.72 g/m^2. Seeding rates were based on recommendations from the seed company and previous work by Elseroad (2001). Weight of slash (limbs and tops) added was 9.1 kg/m^2 fresh weight, creating an approximate slash depth of 0.6 m per plot. This weight and height represented a volume

of slash that was adequate in evenly covering at least 75 % of the plot. Slash diameters were less than 7.62 cm and less than 1 m in length. Only juniper biomass was used for slash treatments for consistency purposes. Juniper species are also the main invading species, therefore targeted in thinning operations.

Soil Stability Methodology

Soil stability (loss or aggradation) was coarsely estimated through erosion bridge measurements (Shakesby 1993, Brockway et al. 2002). At each plot, two permanent stakes 30.5 cm long and 1 m apart were cemented approximately 5 cm above the ground to minimize the movement of stakes. A piece of angle iron was placed on top of the two stakes. We identified and measured the distance from the bridge (angle iron) to the soil surface at three fixed points, equally spaced along the length of the angle iron. It has been suggested that bride data are highly erratic and numerous points per bridge be samples. Due to time constraints only 3 points were measured for each bridge. A 15 cm square with a built-in level was used to precisely measure this distance. We then averaged the three fixed points for estimated soil levels at the plot level. Data on soil stability (soil movement) were summarized for each plot and analyzed for treatment effects, defined as the difference in soil level from the previous year. Baseline soil surface levels were taken in the fall of 2003 across 120 soil erosion bridges.

Field Measurements

We surveyed each of the 120 plots for vegetation and soil movement during August 2003 (pretreatment), August 2004 (post treatment) and August 2005 (post treatment). We identified and estimated foliar cover for each species, soil substrates and

O_i layer (litter) cover using an ocular estimate to the nearest 0.1 %. Plant nomenclature followed the USDA Plants Database (USDA, NRC 2004). In addition, we counted herbaceous species in each plot. In 2005, individual seeded graminoid species were identified for the presence of reproductive inflorescences.

Soil Abiotic Methodology

We collected three soil samples from random locations within each $1m^2$ plot in August 2003 and 2004. Soils were collected to a depth of 10 cm using a 4.2 cm diameter soil core. To avoid seasonal variation, we collected all soils in August 2003 and 2004, concurrently with vegetation sampling. The three soil samples from each plot were then composited together and analyzed for soil abiotic properties and arbuscular mycorrhizal fungal propagules. Soils were passed through a 2 mm sieve prior to chemical analyses. Composite soil samples from each plot were analyzed for pH, total N, organic C, soil moisture and available NO_3-N and NH_4-N. Soil pH was determined in 1:1 soil-water slurry using a pH meter. Total N and organic C were determined using a FLASH EA 1112 Elemental Analyzer. Percent soil moisture was determined gravimetrically and was limited to a single day in 2004. Available NO_3-N and NH_4-N were determined by KCl extraction for freshly collected soil by automated colorimetry using a Technicon Autoanalyzer (Parkinson and Allen 1975). NO_3-N and NH_4-N analyses were only conducted on posttreatment soil samples and limited to a single day in 2004.

Arbuscular Mycorrhizal Fungal Propagule

Bait-plant bioassays were used to quantify the relative amounts of infective propagules of arbuscular mycorrhizal fungi (AMP) within soil samples (Giovannetti and

Mosse 1980, Johnson et al. 1999). Bait-plant bioassay can detect all types of viable mycorrhizal fungal propagules and therefore is considered a more accurate method for quantifying total density of AMP propagules than direct counts spores, or colonized root lengths (Brundrett and Abbott 1994). For each plot, we collected soils to a depth of 10 cm and within 24 hrs, samples were placed into 4 cm diameter by 20 cm deep Conetainers (Stuewe and Sons, Inc., Corvallis, OR, U.S.A.). Corn (*Zea mays*) was used as the bait-plant to determine the amount of infective AMP propagules. We used corn as the host plant, because it grows fast, uniformly and is mycotropic with many AMF species. Corn seeds were germinated, planted in freshly collected soils, and then harvested after 5 weeks or before becoming rootbound within the conetainers. Corn roots were then cut into 2.5 cm segments and randomly subsamples were cleared using a KOH solution and stained in 0.5% Shaeffer Ink (Vierheilin, et al. 1998). Segments of the root length containing AMP structures were quantified by the gridline intersect method using a dissecting microscope (Giovannetti & Mosse 1980).

Microbial Biomass Carbon Methodology

We collected samples for microbial biomass carbon (C) within control and slash treatments in August 2004. Microbial activity is often limited by C availability, so the use of C substrates to assess microbial populations is appropriate (Gallardo and Schleinger 1995). All mineral soil samples for microbial biomass C were collected separately from the other soil samples decribed above. We collected samples at a depth of 10 cm, in August 2004. Soil microbial C was determined using the chloroform ($CHCl_3$) fumigation-extraction method (Brooks et al. 1985, Vance et al. 1987, Haubensak et al. 2002). Approximately 30 g of sieved, field-moist soil was extracted with 100 ml of

0.5 M K_2SO_4 and 30 g of mineral soil was also placed inside a dessicator with a beaker containing 30 ml of $CHCl_3$. The dessicator was repeatedly evacuated to boil the $CHCl_3$ and then left under vacuum for 5 days (Haubensak et al., 2002). After 5 days, the $CHCl_3$ was removed from the soil by repeated evacuations and then the soil subsamples were immediately extracted with 100 ml of 0.5 M K_2SO_4. Extracts were mechanically shaken for one hour, filtered with Whatman #1 filters (pre-leached with deionized water), and frozen until analysis. Organic C concentrations in unfumigated and fumigated extracts were determined by ultraviolet-enhanced persulfate oxidation using a Dohrmann DC-80 Carbon Analyzer with infrared detection (Tekmar-Dohrmann, Cincinnati, OH, USA). Microbial C was calculated by subtracting organic C in the unfumigated extracts from organic C in the fumigated extracts and dividing by a k_{EC} of 0.39 (Sparling et al. 1990).

Statistical Analyses

All data for response variables were summarized as means and variance for each of the four treatments within the two different sites (n =15 for each site). The Shapiro-Wilk test was used to test data for normality and Levene's test was used to test for homoscedasticity of the variance. Significance testing was based on an alpha = 0.05. All analyses were performed using SAS JMP software (SAS Institute 2002).

Soil stability was defined as the change in soil level from the previous year (soil movement). We used analysis of variance (ANOVA) with repeated measures to test treatment, time and treatment by year interactions. Tukey's Honestly Significant Difference (HSD) test was used to make multiple comparisons of all treatment means within each year, following a significant ANOVA result.

Seeded species and soil substrate variables included percent seedling cover, seedling density, percent litter cover and percent soil cover. These data strongly violated the assumption of normality, so we used non-parametric Kruskal-Wallis test to compare individual variables across the four treatments within individual years. Following a statistically significant result, a Mann-Whitney Test was used to make pairwise treatment comparisons. Alpha levels for pairwise comparisons were adjusted by dividing 0.05 by the number of pairwise tests (Bonferonni correction). Repeated measures for seeded species variables within treatments across three years were tested with Wilcoxon signed-ranks tests and following a statistically significant result, we proceeded to Bonferroni-corrected pairwise year comparisons.

All soil abiotic and AMP variables met ANOVA assumptions of normality and equal variance except NO_3-N data and AMP data. Prior to analysis, NO_3-N data and AMP data were square root transformed to meet the normality assumption. We used ANOVA for each nutrient and AMP to compare concentrations among treatments and Tukey's HSD to separate mean concentrations among treatments. A paired t-test was used to assess mean difference of microbial biomass carbon among control and slash only treatments.

Structural Equation Modeling

Structural equation modeling (SEM) is an extension of regression and path analysis that can be used to evaluate hypotheses about complex interacting networks (Bollen 1989, Shipley 2000, Pugesek et al. 2003, Grace 2006). SEM is dependent on the use of theoretically plausible relationships to determine which models are consistent or inconsistent with the multivariate relations in the data (Bollen 1989). When acceptable

models are obtained, the results can potentially indicate the roles that different factors play in a system, and the direct and indirect effects of each factor on a response.

We established an initial structural equation model that represents what we believed to be the most plausible structural relations based on *a priori* knowledge and the limited variables that we collected (Figure 3.1). We acknowledge that not all causal processes and factors that may influence seedling emergence are represented. Six variables were incorporated into the model to explain seedling emergence, slash treatments (slash, no slash), litter cover (% of ground), C:N ratio, NH_4, % soil moisture, and soil particle size. Maximum likelihood estimators and chi-square goodness of fit statistics were used to evaluate model adequacy. Small chi-square values and large *P*-values ($P > 0.05$) indicate that the model-implied covariance structure did not deviate significantly from the observed data. Residuals and modification indices were evaluated to reduce model-data discrepancies. Final structural equation models predicted covariance structures that were consistent with observed data. Analyses were performed using Mplus software (Muthen and Muthen 2005).

We calculated the so-called 'total effects', which are the sum of direct and indirect effects (i.e., standardized path coefficients) from the predictors to one-year seedling emergence. An indirect effect equals the total product of path segments from a predictor to seedling emergence. Total effects are a summary of the strength and sign of the relationship between the factors and seedling emergence.

RESULTS

Soil Stability

Coarse measurements of soil movement in all slash treatments was significantly ($p<0.0001$) lower compared to all non-slash treatments, at both sites (Figure 3.4). Two year mean soil loss among slash treatments at the clay-loam site was 10.0 mm compared to 28.5 mm among non-slash treatments. Mean soil loss in two years among slash treatments at the cinder site was 12.0 mm and 30.3 mm in non-slash treatments. In two years, average soil loss among non-slash treatments was almost 3 times greater than slash treatments. Considerable year to year variation occurred in soil stability. There was a significant decline ($p<.0001$) in soil movement in 2004-2005 compared to 2003-2004 at both sites (Figure 3.4). Average soil loss in 2003-2004 in non slash treatments was 4 and 2 times greater compared to 2004-2005 at the clay-loam and cinder site, respectively. One year following treatments, soil loss in slash treatments was significantly ($p<0.0001$) lower compared to non-slash treatments, at both sites. In 2004-2005, there were accumulations of mineral soil within slash treatments, whereas soil continued to be lost within non-slash treatments. Average soil gained within slash treatments ranged between 0.8 mm and 3.3 mm, while average soil loss within non-slash treatments ranged between 5.5 mm and 9.0 mm.

Litter and Soil Cover

Prior to treatments, no differences were evident in litter cover and mineral soil cover at either the clay-loam or the cinder sites (Table 3.2). Bare soil composed 99% of the intercanopy spaces and litter cover was less than 0.4%. One year following

treatments, litter cover significantly increased within slash treatments compared to non-slash treatments at both the clay-loam and the cinder sites (Table 3.2). Litter cover within slash treatments ranged between 86.3% and 93.6 % and remained approximately less than 1% within non-slash treatments.

Soil Abiotic Properties

In August 2003, prior to treatments, there were no significant differences in pH, total N, and organic C, between treatments within each site (Table 3.3). Soil pH at the clay-loam and cinder sites was slightly alkaline. Organic C values averaged across all treatments at the clay-loam and cinder sites were 0.8 mg/g and 1.03 respectively. Total N averaged less than 0.1 mg/g at each site.

There were no significant changes in soil pH, organic C and total N as a result of treatments (Table 3.3). There were also no significant differences in NH_4-N between treatments (Table 3.3). However, NO_3-N was significantly lower for both the clay-loam site ($p<0.0001$) and cinder site ($p<0.0001$) when comparing slash versus non-slash plots one year after treatments (Figure 3.5).

Soil Biotic Properties

Mycorrhizal fungi colonized all bait-plants within the 2003 and 2004 sampling periods. Prior to treatments, no significant differences in arbuscular mycorrhizal inoculum potential (AMP) were detected between plots, at both sites. On average AMP within the clay-loam and cinder sites were 11.26 % and 11.67 % respectively, before treatment. One year following treatments, AMP was significantly ($p<0.0001$) greater in slash treatments compared to non-slash treatments at both sites (Figure 3.6). Percent root

length colonized with AM fungi in slash treatments was more than two times greater than non-slash treatments at both sites. No statistical differences were evident between slash only and slash and seed treatments, although slash and seed treatments were generally greater in AMP at both sites.

No pretreatment data were collected for microbial biomass C. Samples were only taken from control and slash treatments in 2004. Microbial biomass C was significantly (<0.0001) greater in slash only treatments compared to control treatments at both sites (Figure 3.7).

Seeded Species Response

No differences in total understory cover or density were detected between treatments, prior to treatment at either site. Total understory vegetation was sparse (<3%) at both sites (Table A.2 and A.3). Three out of the four species to be seeded were found in plots at the clay-loam site prior to treatment in Aug. 2003. The combined cover of graminoid species to be seeded at the clay-loam site averaged 0.03 % and the number of individuals was less than 1 plant /m^2 prior to treatment. In the cinder site, no seeded species were detected within plots prior to treatment, though three out of the four species were observed within the area.

One year following seed and slash treatments, foliar cover of seeded species was significantly greater (χ^2=22.3, p<0.0001) in seed and slash treatments compared to the other three treatments, at the clay-loam site (Figure 3.8a). At this site, seeded species cover for control plots increased slightly, whereas seed-only plots showed a 7-fold increase and seed and slash plots increased 13-fold. Two years after treatments were applied, seeded species cover continued to increase significantly (χ^2=26.1, p<0.0001

clay-loam site) in seed and slash treatments compared to the other three treatments, increasing from 0.86 % cover in 2004 to 1.61 % cover in 2005. Seed only treatments showed a slight reduction in foliar cover from 2004 to 2005. Control and slash only treatments showed relatively no improvement between years.

In 2004 at the clay-loam site, the number of seeded species per m^2 also increased significantly (χ^2=27.1, p<.0001) in response to the combination of seed and slash treatments (Figure 3.9a). Similar significant (χ^2=29.3, p<.0001) responses continued in 2005 when comparing seed and slash treatments to the other three treatments. All treatments except slash only decreased in seeded species density when comparing 2004 versus 2005. Species encountered during cover and density measurements were *Elymus elymoides*, *Bouteloua curtipendula* and *Aristida purpurea*. Not all seeded species established equally (Table 3.4). *Elymus elymoides* had the most seedling emergence, whereas *Achnatherum hymenoides* was seeded but not found after treatment.

Treatment effects on seeded species cover and density at the cinder site yielded similar results to the clay-loam site. Mean comparisons for cover were significantly different for pairwise treatment contrasts both in 2004 (χ^2=49.4, p<.0001) and 2005 (χ^2=50.2, p<.0001) (Figure 3.8b). Seeded species foliar cover increased from no seeded species detected in 2003 to an average of 3.8 % in seed and slash treatments, whereas seed only treatments increased less than 0.1 %, one year following the application of treatments.

Seedling density per m^2 was also significantly different in 2004 (χ^2=49.3, p<.0001) and 2005 (χ^2=47.6, p<.0001) when comparing across treatments. Seedlings increased from no individuals detected in 2003 to an average of 46.6 individuals/m^2 on

seed and slash treatment plots and 2 individuals/m^2 on seed only plots in 2004 (Figure 3.9b). No seeded species were detected within control plots in 2004 at the cinder site. Almost a two fold reduction of individuals/m^2 and a 0.1% decrease in foliar cover was detected in seed and slash plots when comparing 2004 versus 2005. Seed only plots showed a slight increase in individuals/m^2 and foliar cover when comparing 2004 to 2005. Slash only treatments increased from no cover in 2004 to an average of .8% cover in 2005. Not all species that were seeded within treatments emerged after two growing seasons (Table 3.5). Once again, *Achnatherum hymenoides* was not evident during the post treatment measurement.

Across both sites, 42% of the seed and slash plots had at least one seeded species that was reproductively mature, whereas 7 % of the seed only plots contained a reproductively mature seeded species (illustrated in Figure. A.1).

Structural Equation Modeling

The *a priori* structural equation model (Fig. 3.1) was found to be consistent with the data ($\chi^2 = 2.2$, df = 7, $P = 0.95$). However, a few variables were found to be nonsignificant paths ($p > 0.05$) and therefore were removed to simplify the model. Soil particle size was found to be nonsignificant and was dropped from the model. We also removed slash treatment direct paths to seedling emergence and NH_4. The effects of slash treatment on seedling emergence resulted in indirect paths through associations with litter and soil moisture. The effects of slash treatment on NH_4 also resulted in an indirect path through associations with C:N. The resulting model was found to not differ significantly from the data ($\chi^2 = 4.0$, df = 8, $P = 0.86$) and explained 46% of the variation

in seedling emergence, 4% of the variation in C:N, 12% of the variation in soil moisture, and 99% of the variation in litter accumulations (Fig. 3.10).

The 'total effects' and standardized path coefficients for direct and indirect effects on seedling emergence are presented in Table 3.6. Slash treatments had positive total effects on seedling emergence (.54), which were entirely indirect through associations with litter and soil moisture. Litter had strong positive effects on seedling emergence (.47). NH_4 and soil moisture both had positive total effects (.32) and (.20) respectively, on seedling emergence. C:N had a weak positive total effect (.11) on seedling emergence, which was entirely indirect through association with NH_4.

DISCUSSION

Scattering slash across plots produced an immediate increase in litter cover that led to a decrease in soil exposure at both sites. Both increases in litter cover and the role of residual woody debris had an apparent impact on soil loss within interspaces. Soil loss after two years was on average 276 % less in slash treatments compared to non-slash treatments across both sites. Other studies also found obvious decreases in sediment yields following slash treatments (Evans 1988, Hasting et al. 2003). In contrast, Brockway et al. (2002) found no differences in sediment loss when comparing slash treatments to controls located within intercanopy spaces. Soil erosion can be highly dependent on variables such as type of precipitation, slope, scale of monitoring, the amount of protective cover on the soil surface, and soil taxonomy (Wilcox and Breshears 1995, Hasting et al. 2003). In our study, slopes within interspaces were less than 5 %. The scale at which we measured the rate of soil runoff was at the intercanopy level and in two years we observed an average 11.01 mm and 29.4 mm of soil loss in slash and non-

slash treatments, respectively. These values are considerably higher when compared to other studies. Jacob et al. 2002 found average soil loss after two years, using similar treatments to be 7 mm and 12 mm in slash and non-slash treatment, respectively. Methods for measuring soil erosion can be highly erratic and numerous points should be sampled to control for extreme variation. Methods and sampling differences could possible explain the difference in magnitude between the two studies. We also found rates of soil loss to be highly variable from year to year. In 2004, there was over four times more sediment loss in non-slash plots compared to 2005. The reduction of summer precipitation in 2005 as compared to 2004 and an increase in total vegetation may explain this variation. Wood et al. (1987) explained that large bare, connected interspaces have a much lower infiltration rate than neighboring tree mounds therefore becoming major pathways for sediment runoff. This could explain the large amount of soil loss, even after implementing slash treatments. Although sediment runoff continued after treatments, residual woody and litter debris appeared to provide favorable conditions for microbial activity and graminoid seedling emergence and establishment.

One year after applying slash treatments, juniper needles began to decompose and new soil aggregates were observed around the juniper needle cast. These newly observed soil aggregates could be explained by increases in microbial activity, and the protection of increased litter cover guarding against the impact of raindrops, allowing soil particles to bind together. We found an average of 271.6 mg/g of microbial biomass C across all slash treatment. This was more than 1.5 times greater than control plots. Slash left in place has shown to increase organic C availability, leading to increases in microbial biomass (Lundgen 1982). Other studies have concluded that soil microbial populations

will be higher where soil resources are higher (Lundgen 1982, Herman et al. 1995, Murphy et al. 1998). Soil chemical properties exhibited no significant changes one year after treatments were applied. However, slight increases were noticeable for organic C. Small differences in organic C can greatly affect the distribution and structure of pores within the soil influencing the storage and movement of water and thereby facilitating microbial activity (Perry et al. 1987, Wardle 1992). Increased amounts of moisture and carbon substrates can promote increased rates of microbial respiration, thereby regulating nutrient cycling and promoting vegetation growth and establishment (Berg et al. 1982, Gallardo and Schlesinger 1995, Murphy et al. 1998). The availability of soil nutrients is generally controlled by the below ground biota, which regulates the mineralization and immobilization of nutrients (Schlesinger et al. 1990). We found significant decreases in NO_3-N in slash treatments compared to non-slash treatments. This decrease may be related to the immobilization of NO_3-N through increased microbial activity (Schlesinger et al. 1990).

We also found significantly higher AMP colonization in slash treatments compared to non-slash treatments. Over all average AMP colonization was 14.3 % and 6.5 % in slash and non-slash treatments, respectively. Numerous researchers have suggested that the recovery time of disturbed ecosystems is highly dependent upon the establishment of AM fungi (Reeves et al. 1979, Allen and Allen 1980, Perry et al. 1987, Korb et al. 2004). The severity of the disturbance can greatly influence the abundance of resistant AM fungi. However, resistant mycorrhizael propagules can be proficient infectors and account for high levels of colonization even after intense disturbances (Allen and Allen 1980, Korb et al 2004).

Seeding response to slash treatments was clear when compared to seed only treatments, 24 months following application of treatments. Our results indicate that the addition of both slash and seed treatments generated higher percent cover, more seedling emergence and more reproductively producing graminoids compared to the other treatments. These results were consistent with other studies, indicating the necessity to modify the bare soil through the addition of slash to effectively increase plant establishment (Chong 1994, Jacob and Gatewood 1999, Brockway et al. 2002). Furthermore, without the addition of seeds, there would have been little to no new grass establishment. In our study, graminiod cover remained low two years after slash and seed treatments (1.6 % and 3.8 % depending on site). However, seeded species accounted for 19 % of the total cover (14.3 %) and 49 % of the total density across all slash and seed treatments, which substantially affected the overall net increase in graminoid cover, and density in 2005. Graminiod seeded species clearly responded best when slash was scattered on the plot, thereby demonstrating that slash amendments do create favorable habitats for enhancing establishment and potential long term survival of native graminoid species.

Only one species established well from seed, western bottlebrush. Establishment of western bottlebrush accounted for greater than 70 % of the total graminoid cover in slash and seed treatments. Elseroad et al. (2005) also found western bottlebrush to be highly successful in establishing on a road rehabilitation project. Sideoats grama accounted for just over 20 % of the total graminoid cover. Low establishment for the two other grass species purple threeawn and Indian ricegrass may have resulted from unmet seed dormancy or scarification requirements.

The SEM model results suggest a variety of plausible mechanisms whereby slash treatments can directly and indirectly influence one year seedling emergence. Approximately half of the observed variation in seedling emergence appears to be related to positive effects of available NH_4, moisture % and litter cover. This suggest that soils with higher available NH_4, moisture % and litter cover can sustain greater seedling emergence. Available NH_4, and moisture % were only measure for one specific day and showed no significant differences due to treatments, yet they contributed unique explanatory values to our exploratory model. Consistent with earlier studies, microsites associated with high moisture and humidity can sustain greater plant emergence (Harper et al. 1965, Young and Evan 1987, Chambers 2000). Small changes in soil moisture can greatly influence seedling germination and emergence (Harper et al. 1965). Litter cover which was highly correlated with slash treatments, was found to have the strongest association with seedling emergence (standardized coefficient value of .47). Positive relationship between litter cover and seedling emergence was also found in other studies (Fowler 1988, Call and Roundy 1991, Jacobs and Gatewwod 1999). There were no significant differences in C:N ratios due to treatments, however they contributed significantly to our model. C:N ratios had a indirect effect on seedling emergence in which case C:N levels influenced the amount of available NH_4. It has been suggested that the quality of litter can influence C:N ratio, allowing more available nitrogen to be readily available for plant uptake (Murphy et al. 1998). Overall, all factors within this model had a positive effect on seedling emergence. In most cases, the bivariate relations between abiotic conditions and seedling emergence were non-significant. However, once the covariation among abiotic predictors was controlled in the context of the multivariate

model, all predictors significantly affected seedling emergence. This result illustrates the capacity of SEM to reveal masked and suppressed relationships within multivariate space (Grace and Pugesek 1998).

MANAGEMENT IMPLICATIONS

Recovery of native vegetation in pinyon-juniper woodlands are driven by management objectives which may include enhancement of native ecosystem structure, composition and ecosystem processes. Revegetation objectives and practices must be grounded by ecological and economic realities or they will not be implemented. Realization that climatic factors such as low precipitation and extreme temperature variation are the norm for many pinyon-juniper woodlands exerts an overriding effect on whether a revegetation project succeeds or fails. We implemented an amelioration treatment by utilizing residual woody debris to improve recovery of native graminoid species specifically in the interspaces of mature pinyon-juniper woodlands. Our objectives were to create favorable microsites to enhance long-term herbaceous establishment. Conditions that can improve water and nutrient availability, plus trap and retain seeds have consistently shown to increase seedling establishment across different ecosystems (Fowler 1986, Tongway and Ludwig 1996a Chambers 2000). Therefore, in order to maximize the effectiveness of seedling recruitment whether it is artificial or natural recruitment, favorable soil amendment should be considered.

The early results suggest several implications for establishing herbaceous species in pinyon-juniper woodlands of the southwest.

- When recovering native vegetation, seeding and scattering on site mulching material is far more effective in increasing plant cover and density than seeding

alone. However, if persistent seedbank and reproductive viable vegetation occur on site, thinning and scattering woody debris may be adequate in augmenting native vegetation.

- Scattering slash across interspaces may help retain essential soil nutrients and establish important plant-soil interaction, which can contribute significantly to the potential long-term development of native vegetation. Monitoring the soil microbial community can provide useful information about the health and stability of an ecosystem.
- Not all seeded species established. *Elymus elmoides* and *Bouteloua curtipedula* had the best response when seeded in these harsh environments.
- Evaluate each site on a case by case basis. Soil physical characteristics such as substrate composition seem to contribute to the success of vegetation establishment
- Slash mulch treatments should be considered a temporary solution to aid the initial recovery of vegetation within pinyon juniper woodlands. The establishment of understory vegetation should be considered the primary goal for the long-term recovery of these degraded pinyon juniper ecosystems.

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Table 3.1. Seeded species and seeding rate applied to seeded plots.

Functional group	Ps Pathway	Seeded Species	Common Name	Seeding rate (g/m2)
Graminoid	C4	*Aristida purpurea*	purple threeawn	2.43
	C3	*Achnatherum hymenoides*	Indian ricegrass	2.43
	C4	*Bouteloua curtipendula*	blue grama	2.43
	C3	*Elymus elymoides*	western bottle-brush grass	2.43
			Total	**9.72**

Table 3.2. Percent litter and soil cover under different treatments in 2003 and 2004, pretreatment and 12 months post treatment represented at two different sites.

	Clay-Loam			
	Control	**Seed**	**Slash**	**Slash + Seed**
Litter Cover %				
2003	**0.55a** (0.50)	**0.28a** (0.27)	**0.28a** (0.14)	**0.07a** (0.04)
2004	**2.20a** (1.03)	**1.17a** (0.68)	**86.15b** (2.50)	**88.31b** (1.71)
Mineral Soil %				
2003	**99.36a** (0.40)	**99.60a** (0.27)	**99.63a** (0.14)	**99.85a** (0.04)
2004	**97.69a** (1.03)	**98.71a** (.69)	**13.77b** (2.50)	**11.37b** (1.71)
	Cinder			
	Control	**Seed**	**Slash**	**Slash + Seed**
Litter Cover %				
2003	**0.21a** (0.14)	**0.19a** (0.13)	**0.40a** (0.22)	**0.58a** (0.32)
2004	**1.00a** (0.50)	**0.60a** (0.27)	**89.80b** (2.37)	**92.63b** (0.85)
Mineral Soil %				
2003	**99.88a** (0.14)	**99.73a** (0.13)	**99.54a** (0.22)	**99.34a** (0.32)
2004	**99.92a** (0.50)	**99.31a** (.27)	**10.11b** (2.37)	**6.40b** (0.85)

Data expressed as **means** (SE) (n = 15). Within each row, values indexed by a different letter are significant difference at $\alpha = 0.05$.

Table 3.3. Soil abiotic properties under different treatments in 2003 and 2004, pretreatment and 12 months post treatment represented at two different sites.

	Clay-Loam				Cinder			
	Control	**Seed**	**Slash**	**Slash + Seed**	**Control**	**Seed**	**Slash**	**Slash + Seed**
Organic C (g/kg)								
2003	**8.4** (0.5)	**8.0** (0.5)	**7.9** (0.4)	**7.5** (0.4)	**10.8** (0.9)	**9.2** (0.7)	**10.6** (0.7)	**10.5** (0.6)
2004	**6.5** (0.5)	**0.70** (.0.5)	**7.8** (0.5)	**8.0** (0.5)	**9.2** (0.9)	**9.1** (0.9)	**10.8** (0.7)	**10.7** (0.6)
Total N (g/kg)								
2003	**0.8** (0.03)	**0.8** (0.02)	**0.8** (0.02)	**0.7** (0.03)	**0.7** (0.07)	**0.6** (0.04)	**0.7** (0.05)	**1.2** (0.5)
2004	**0.6** (0.04)	**0.6** (0.02)	**0.6** (0.03)	**0.7** (0.03)	**0.8** (.07)	**0.7** (0.06)	**0.8** (0.05)	**0.9** (0.04)
C:N								
2003	**10.60** (.25)	**10.56** (.36)	**10.53** (.22)	**10.35** (.22)	**14.60** (.29)	**14.32** (.29)	**14.54** (.43)	**14.48** (.40)
2004	**10.83** (.44)	**10.88** (.45)	**10.98** (.35)	**11.61** (.33)	**11.96** (.35)	**11.95** (.36)	**12.60** (.25)	**12.34** (.24)
NH_4-N (mg/g)								
2004	**0.96** (.08)	**1.01** (.06)	**0.78** (.07)	**0.79** (.09)	**1.62** (.64)	**0.89** (.08)	**0.98** (.08)	**1.15** (.13)
pH								
2003	**7.45** (.10)	**7.50** (.13)	**7.60** (.13)	**7.60** (.09)	**7.93** (.08)	**7.83** (.06)	**7.93** (.08)	**7.85** (.05)
2004	**7.44** (.21)	**7.29** (.26)	**7.51** (.20)	**7.70** (.20)	**7.51** (.08)	**7.39** (.14)	**7.74** (.14)	**7.85** (.03)

Data expressed as **means** (SE) for each soil chemical property (n = 15). No significant difference at $\alpha = 0.05$.

Table 3.4. Density (# plants) per m^2 and percent cover for each seeded species under different treatments in 2003, 2004 and 2005, pretreatment, 12 months and 24 months post treatment at the Clay-Loam Site.

Species	Photosynthetic Pathway	Avg. Cover (%) 2003	Avg. Cover (%) 2004	Avg. Cover (%) 2005	Avg. Density (m^2) 2003	Avg. Density (m^2) 2004	Avg. Density (m^2) 2005
Elymus elymoides	C3						
Control		0	**0.01** (0.01)	0	0	**0.20** (0.20)	0
Seed		0	**0.13** (0.13)	**0.07** (0.07)	0	**1.73** (1.46)	**0.02** (0.02)
Slash		0	**0.06** (0.05)	**0.26** (0.13)	0	**0.40** (0.24)	**1.80** (0.83)
Slash + Seed		0	**0.86** (0.30)	**1.10** (0.29)	0	**7.27** (1.74)	**4.90** (1.02)
Bouteloua curtipendila	C4						
Control		0	0	0	0	0	0
Seed		0	**0.12** (0.08)	**0.20** (0.12)	0	**0.27** (0.21)	**0.53** (0.31)
Slash		0	0	0	0	0	0
Slash + Seed		0	0	**0.52** (.21)	0	0	**0.93** (0.27)
Artistida purpurea	C4						
Control		**0.02** (0.02)	**0.05** (0.05)	**0.06** (0.06)	**0.13** (0.13)	**0.13** (0.13)	**0.13** (0.13)
Seed		**0.09** (0.09)	**0.03** (0.03)	**0.01** (0.01)	**0.20** (0.20)	**0.14** (0.14)	**0.07** (0.07)
Slash		**0.02** (0.02)	**0.02** (0.02)	**0.02** (0.02)	**0.07** (0.07)	**0.07** (0.07)	**0.07** (0.07)
Slash + Seed		0	0	0	0	0	0
Achnatherum hymenoides	C3						
Control		0	0	0	0	0	0
Seed		0	0	0	0	0	0
Slash		0	0	0	0	0	0
Slash + Seed		0	0	0	0	0	0

Data expressed as **means** (SE) (n = 15).

Table 3.5. Density (# plants) per m^2 and percent cover for each seeded species under different treatments in 2003, 2004 and 2005, pretreatment, 12 months and 24 months post treatment at the Cinder Site.

Species	Photosynthetic Pathway	Avg. Cover (%) 2003	2004	2005	Avg. Density (m^2) 2003	2004	2005
Elymus elymoides	C3						
Control		0	0	**0.01** (0.01)	0	**0.07** (0.07)	0
Seed		0	**0.07** (0.03)	**0.11** (0.03)	**1.67** (0.68)	**3.2** (1.46)	**1.67** (0.68)
Slash		0	0	**0.84** (0.17)	0	**7.87** (1.25)	0
Slash + Seed		0	**3.88** (0.62)	**2.83** (0.60)	**46.6** (6.72)	**24.5** (3.41)	**46.6** (6.72)
Bouteloua curtipendila	C4						
Control		0	0	0	0	0	0
Seed		0	**0.01**(.01)	**0.02** (0.02)	**0.07** (0.07)	**0.07** (0.07)	**0.07** (0.07)
Slash		0	0	0	0	0	0
Slash + Seed		0	0	**0.67** (0.31)	0	**2.0** (0.76)	0
Artistida purpurea	C4						
Control		0	0	0	0	0	0
Seed		0	**0.01** (0.01)	**0.01** (0.01)	**0.27** (0.21)	**0.07** (0.07)	**0.27** (0.21)
Slash		**0.02** (0.02)	**0.02** (0.02)	**0.02** (0.02)	0	0	0
Slash + Seed		0	0	**0.01** (0.01)	0	**0.13** (0.09)	0
Achnatherum hymenoides	C3						
Control		0	0	0	0	0	0
Seed		0	0	0	0	0	0
Slash		0	0	0	0	0	0
Slash + Seed		0	0	0	0	0	0

Data expressed as **means** (SE) (n = 15).

Table 3.6. Standardized path coefficients direct, indirect and total effects on graminoid species emergence (1-year establishment).

Factors	Direct	Indirect	Total
Slash Treatments		0.54	0.54
Litter	0.47		0.47
NH_4	0.32		0.32
Moisture	0.20		0.20
C:N		0.11	0.11

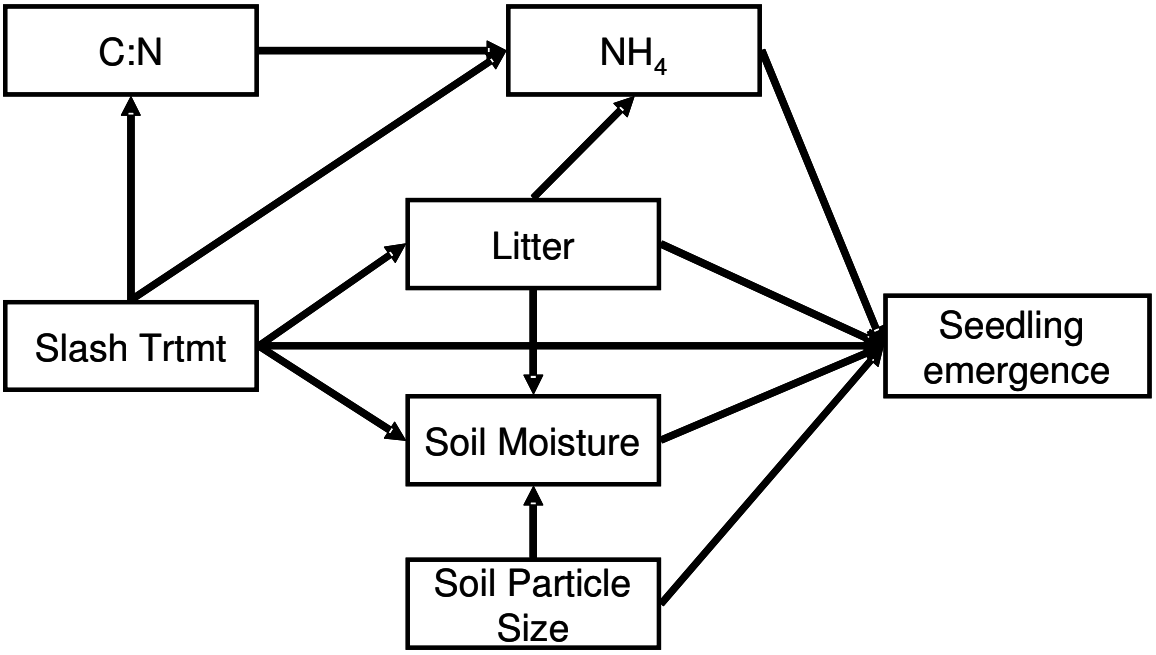

Figure 3.1. *A priori* structural equation model.

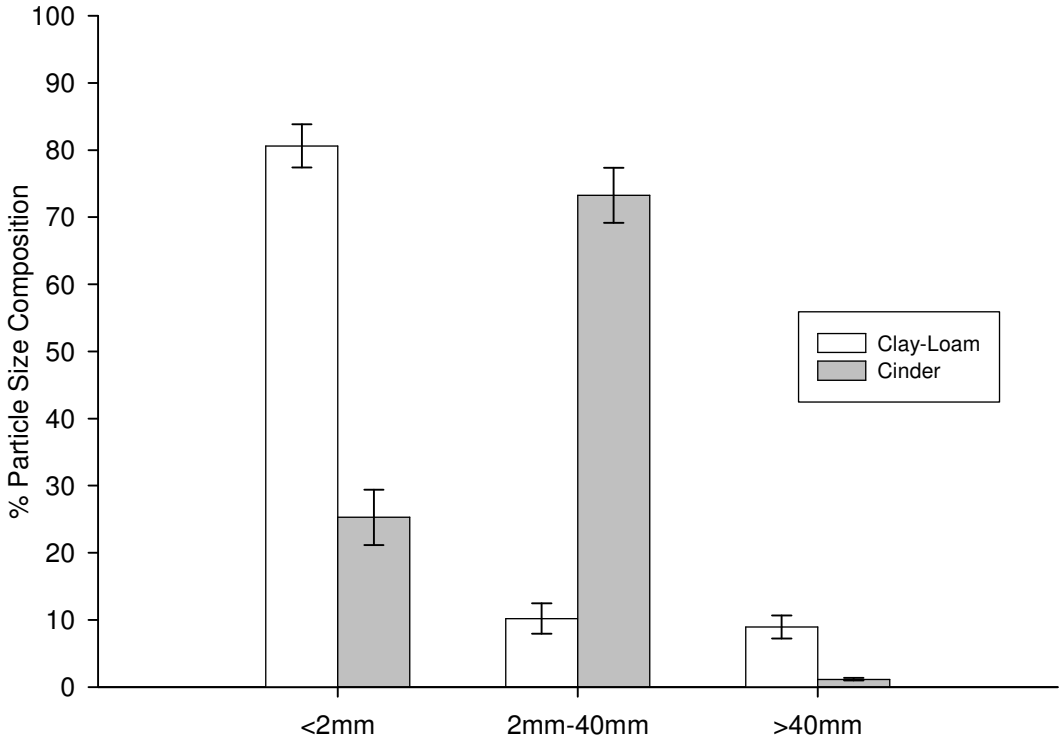

Figure 3.2. Average percent cover of soil particle size at each site. Bars represent 1 standard error (n = 60).

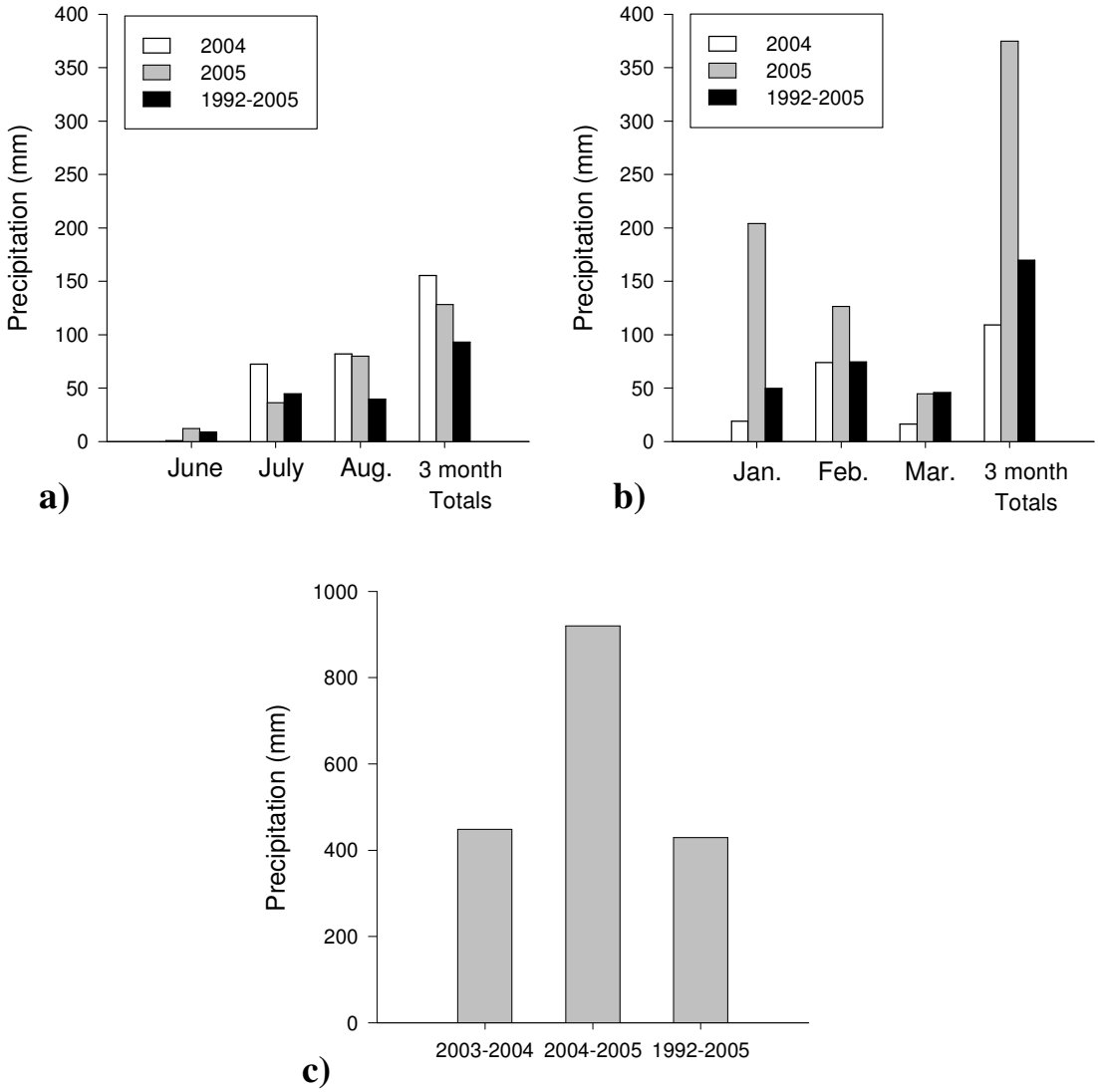

Figure 3.3. Monthly precipitation in 2004 and 2005 versus 13-year average. Summer (June-September) averages are shown in **a.** and winter (December-March) averages are shown in **b.** Annual precipitation (2003-2005, August –June) verse 13-year average (1992-2005) shown in **c**.

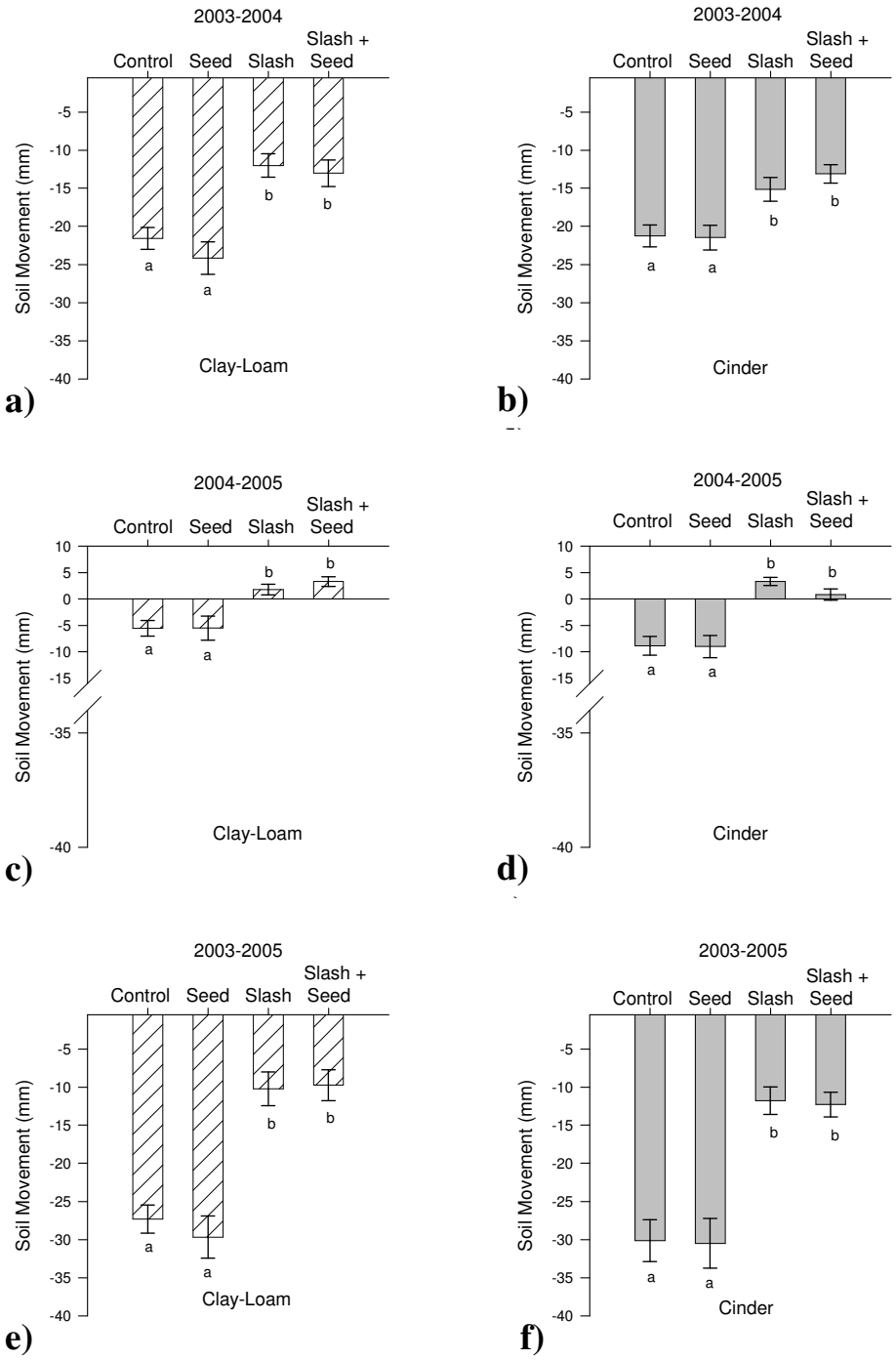

Figure 3.4. Average soil movement (mm) under experimental treatments at two different sites for 2003-2004, 2004-2005 and 2003-2005. Different letters index a significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15).

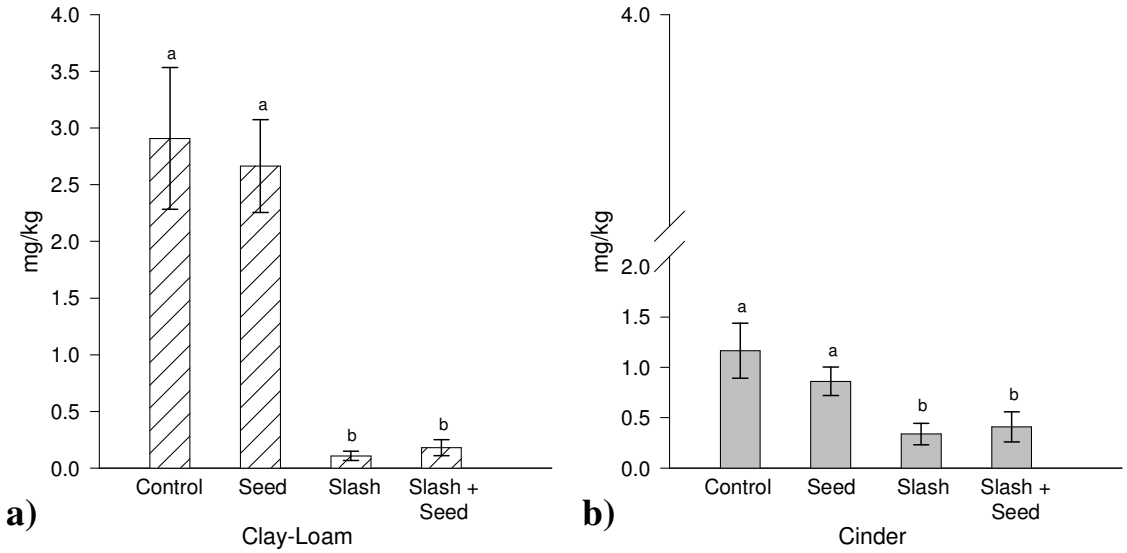

Figure 3.5. Average NO_3-N (mg/kg) under experimental treatments at two different sites in 2004. Different letters index a significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15).

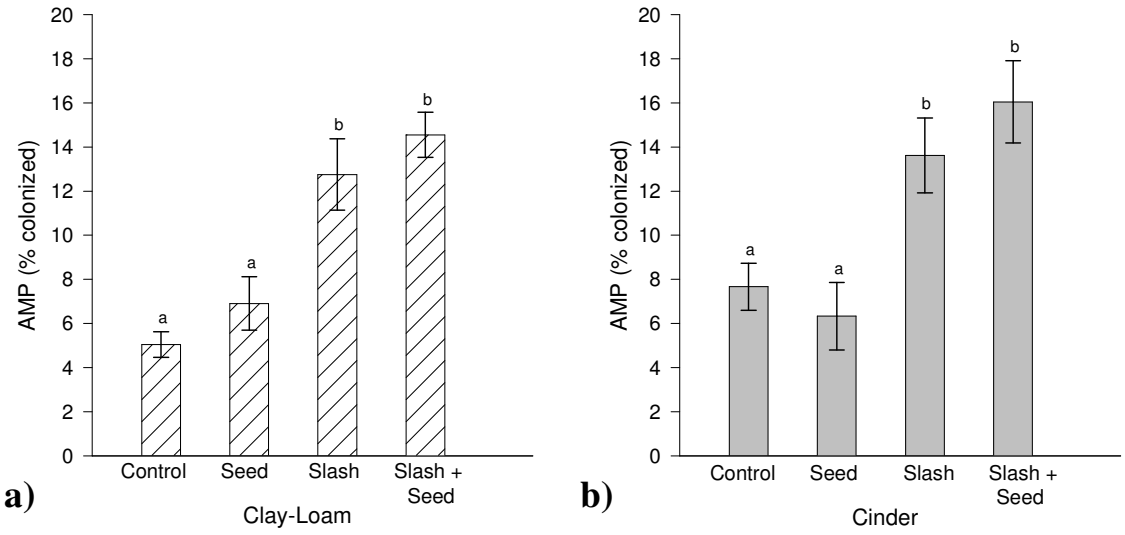

Figure 3.6. Average relative root colonization with arbuscular mycorrhizal fungi under experimental treatments at two different sites in 2004. Different letters index a significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15).

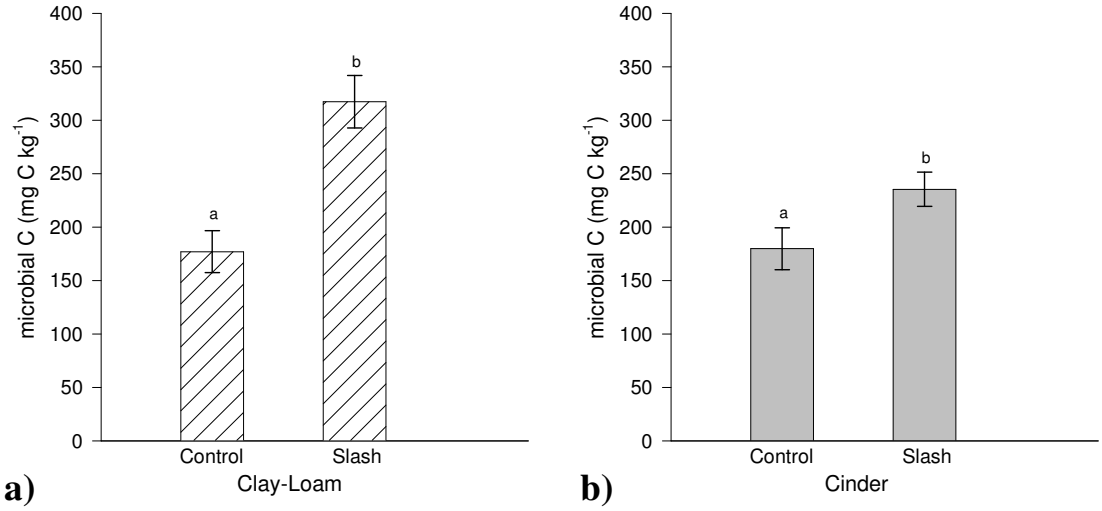

Figure 3.7. Average microbial carbon (mg/kg) under experimental treatments at two different sites in 2004. Different letters index a significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15).

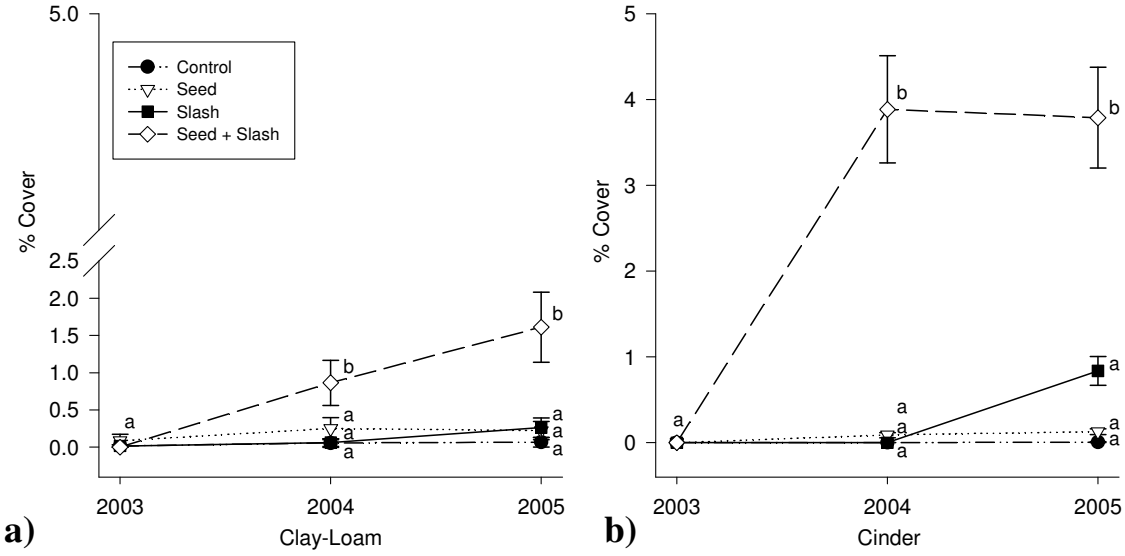

Figure 3.8. Average percent cover for seeded graminoids under experimental treatments in 2003, 2004, 2005, pretreatment, 12 months and 24 months post treatment, represented at two different sites. Values indexed within each year by a different letter are significantly different at $\alpha = 0.05$.

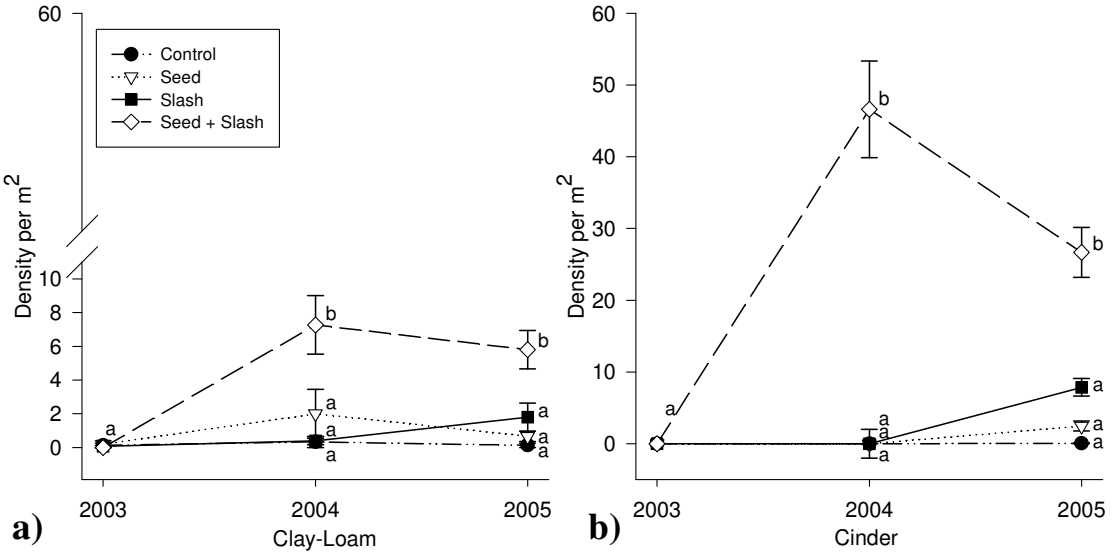

Figure 3.9. Average percent cover for seeded graminoids under experimental treatments in 2003, 2004, 2005, pretreatment, 12 months and 24 months post treatment, represented at two different sites. Values indexed within each year by a different letter are significantly differenct at $\alpha = 0.05$.

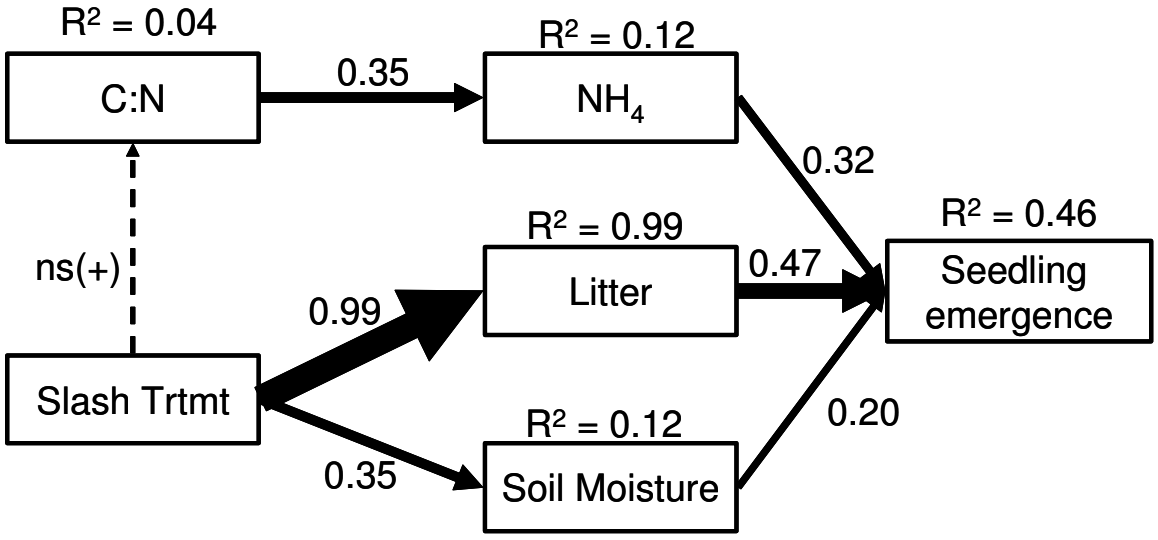

Figure 3.10. Final structural equation model with standardized path coefficients (χ^2 = 4.0, df = 8, *P* = 0.86).

Table A.1. List of species found within study plot at each site. Nomenclature based on USDA Plant Database (USDA, NRCS 2004).

Clay-Loam Site				Cinder Site			
Functioinal groups		**Species**	**Common Name**	**Functioinal groups**		**Species**	**Common Name**
Annual		*Cordylanthus parviflorus*	purple bird's beak	Annual		*Chenopodium album*	lambsquarters
		Polygonum douglasii	Douglas' knotweed			*Epilobium brachycarpum*	tall annual willowherb
		Portulaca oleracea	little hogweed			*Eriogonum davidsonii*	Davidson's buckwheat
Exotic		*Lactuca serriola*	prickly lettuce			*Nicotiana attenuata*	coyote tobacco
Graminoid	C4	*Aristida purpurea*	purple threeawn			*Polygonum douglasii*	Douglas' knotweed
	C4	*Bouteloua curtipendula*	blue grama			*Portulaca oleracea*	little hogweed
	C4	*Bouteloua gracilis*	sideoats grama	Exotic	C3	*Bromus tectorum*	cheatgrass
	C3	*Elymus elymoides*	western bottle-brush grass			*Lactuca serriola*	prickly lettuce
	C3	*Hesperostipa comata*	needle & thread	Graminoid	C4	*Aristida purpurea*	purple threeawn
N fixer		*Psoralidium tenuiflorum*	slimflower scurfpea		C4	*Bouteloua gracilis*	blue grama
Perennial		*Agoseris glauca*	pale agoseris		C3	*Elymus elymoides*	western bottle-brush grass
		Arabis fendleri	Fendler's rockcress	N fixer		*Lotus uthahensis*	Utah birdsfoot trefoil
		Chaenactis douglasii	Douglas' dustymaiden			*Lupinus kingii*	King's Lupine
		Chamaesyce albomarginata	whitemargin sandmat			*Phaseolus angustissium*	slimleaf bean
		Eriogonum corymbosum	crispleaf buckwheat			*Psoralidium tenuiflorum*	slimflower scurfpea
		Hymenopappus filifolius	fineleaf hymenopappus	Perennial		*Chaenactis douglasii*	Douglas' dustymaiden
		Penstemon linarioides	toadflax penstemon			*Chamaesyce albomarginata*	whitemargin sandmat
		Sphaeralcea parvifolia	smallflower globemallow			*Eriogonum corymbosum*	crispleaf buckwheat
Shrub		*Purshia mexicana*	Mexican cliffrose			*Eriogonum umbellatum*	sulphur wildbuckwheat
						Hymenopappus filifolius	fineleaf hymenopappus
						Hymenoxys richardsonii	pingue hymenoxys
						Ipomopsis aggregata	skyrocket gilia
						Machaeranthera canescens	hoary aster
						Packera multilobata	lobeleaf groundsel
						Penstemon linarioides	toadflax penstemon
						Penstemon virgatus	upright blue beardtongue
						Sphaeralcea parvifolia	smallflower globemallow
						Townsendia incana	hoary townsendia
				Shrub		*Artemisia tridentata*	big sagebrush

Table A.2. Average percent cover of functional group species within treatment for Clay-loam Site. Standard error in parenthesis (n = 15 within each site). Percent of total understory species cover for each functional group.

Year	Treatment	Total	Annual	Exotic	Graminoid	N fixer	Perennial	Shrub
2003	Control	**2.48** (0.91)	**1.11** (0.48) 44.8%	0	**0.02** (0.02) 0.8%	**0.85** (0.66) 34.3%	**0.01** (0.01) 0.4%	**0.49** (0.18) 19.8%
	Seed	**1.55** (1.67)	**0.73** (0.34) 47.1%	0	**0.09** (0.09) 5.8%	**0.50** (0.16) 32.3%	**0.14** (0.11) 9.0%	**0.09** (0.05) 5.8%
	Slash	**2.40** (0.92)	**1.11** (0.72) 45.8%	0	**0.02** (0.02) 0.8%	**1.00** (0.54) 41.7%	0	**0.28** (0.14) 11.7%
	Seed + Slash	**1.57** (0.36)	**0.67** (0.26) 42.7%	0	0	**0.68** (0.34) 43.3%	**0.01** (0.01) 0.6%	**0.21** (0.10) 13.4%
2004	Control	**4.49** (0.94)	**2.55** (0.96) 56.8%	0	**0.07** (0.06) 1.6%	**0.29** (0.67) 6.5%	**0.43** (0.25) 9.6%	**1.15** (0.25) 25.6%
	Seed	**3.50** (1.07)	**2.00** (0.94) 57.1%	0	**0.29** (0.15) 8.3%	**0.04** (0.02) 1.1%	**0.52** (0.40) 14.9%	**0.65** (0.30) 18.6%
	Slash	**1.66** (0.57)	**0.36** (0.21) 21.7%	0	**0.10** (0.06) 6.0%	**0.72** (0.36) 43.4%	**0.13** (0.13) 7.8%	**0.35** (0.16) 21.1%
	Seed + Slash	**2.10** (0.37)	**0.11** (0.04) 5.2%	0	**0.87** (0.30) 41.4%	**0.81** (0.28) 38.6%	**0.19** (0.17) 9.0%	**0.12** (0.07) 5.7%
2005	Control	**16.8** (2.79)	**4.82** (1.14) 28.7%	0	**0.10** (0.07) 0.6%	**9.85** (2.31) 58.6%	**0.41** (0.27) 2.4%	**1.62** (0.49) 9.6%
	Seed	**10.96** (2.95)	**3.13** (1.31) 28.6%	0	**0.24** (0.12) 2.2%	**6.27** (2.05) 57.2%	**0.71** (0.54) 6.5%	**0.61** (0.24) 5.6%
	Slash	**11.64** (2.41)	**4.14** (1.13) 35.6%	**0.01** (0.01) 0.1%	**0.26** (0.13) 2.2%	**6.32** (2.24) 54.3%	**0.36** (0.25) 3.1%	**0.55** (0.24) 4.7%
	Seed + Slash	**16.34** (2.15)	**6.24** (1.43) 38.2%	0	**1.63** (0.48) 10.0%	**8.00** (1.69) 49.0%	**0.13** (0.09) 0.8%	**0.34** (0.14) 2.1%

Table A.3. Average percent cover of functional group species within treatment for Cinder Site. Standard error in parenthesis (n = 15 within each site). Percent of total understory species cover for each functional group.

Year	Treatment	Total	Annual	Exotic	Graminoid	N fixer	Perennial	Shrub
2003	Control	**1.28** (0.49)	0	0	0	0	**0.53** (0.46)	**0.75** (0.28)
							41.4%	58.6%
	Seed	**0.35** (.017)	0	0	0	**0.01** (0.01)	**0.05** (0.05)	**0.29** (0.17)
						2.9%	14.3%	82.9%
	Slash	**0.27** (.011)	0	0	0	**0.04** (0.03)	**0.11** (0.08)	**0.12** (0.05)
						14.8%	40.7%	44.4%
	Seed + Slash	**0.36** (.012)	0	0	**0.01** (0.01)	0	**0.17** (0.08)	**0.18** (0.06)
					2.8%		47.2%	50.0%
2004	Control	**3.51** (1.33)	**0.12** (0.08)	0	0	**0.68** (0.67)	**1.89** (1.03)	**0.82** (0.24)
			3.4%			19.4%	53.8%	23.4%
	Seed	**1.46** (.071)	**0.12** (0.07)	**0.15** (0.13)	**0.09** (0.03)	**0.37** (0.28)	**0.39** (0.25)	**0.34** (0.13)
			8.2%	10.3%	6.2%	25.3%	26.7%	23.3%
	Slash	**2.55** (1.25)	**0.01** (0.01)	0	0	**1.26** (1.20)	**0.98** (0.58)	**0.30** (0.14)
			0.4%			49.4%	38.4%	11.8%
	Seed + Slash	**6.84** (1.30)	**0.02** (0.02)	**0.13** (0.13)	**3.92** (0.61)	**1.12** (0.93)	**1.03** (0.52)	**0.62** (.27)
			0.3%	1.9%	57.3%	16.4%	15.1%	9.1%
2005	Control	**8.02** (3.10)	**0.89** (0.87)	0	**0.01** (0.01)	**1.48** (1.33)	**4.89** (2.09)	**0.75** (0.28)
			11.1%		0.1%	18.5%	61.0%	9.4%
	Seed	**3.59** (1.04)	**0.15** (0.10)	0	**0.13** (0.03)	**1.43** (0.75)	**1.34** (0.55)	**0.54** (0.25)
			4.2%		3.6%	39.8%	37.3%	15.0%
	Slash	1**0.67** (3.02)	**0.41** (0.22)	**0.25** (0.23)	**0.84** (0.17)	**6.93** (2.64)	**1.98** (0.89)	**0.26** (0.11)
			3.8%	2.3%	7.9%	64.9%	18.6%	2.4%
	Seed + Slash	**12.34** (3.10)	**0.13** (0.06)	**0.04** (0.03)	**3.74** (0.60)	**5.43** (2.91)	**2.19** (0.82)	**0.81** (0.36)
			1.1%	0.3%	30.3%	44.0%	17.7%	6.6%

Figure A.1. Picture of before treatment (August 2003) and post treatment (August 2005).

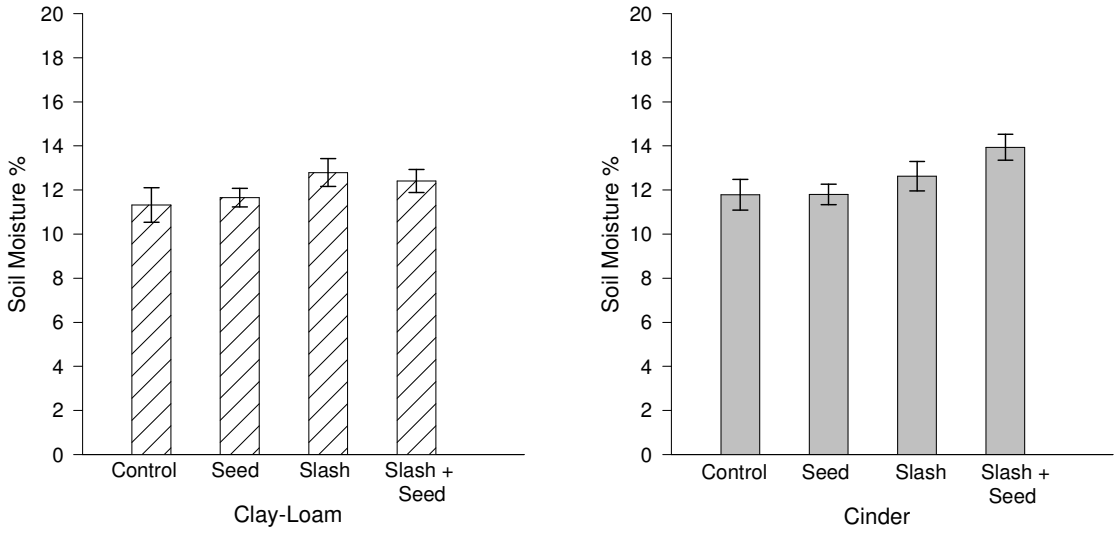

Figure A.2. Average percent soil moisture under different treatments at two different sites. No significant difference between treatments at $\alpha = 0.05$ within each site.

CHAPTER 4

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Summary

The objective of this research was to create favorable habitats for enhancing the establishment and survival of perennial plants in degraded pinyon-juniper woodlands. Degradation of arid and semi-arid ecosystems is a world-wide phenomenon. The more degraded the land, the greater the management and cost required to return the system to a desired state (Hobbs and Norton 1996). In order to reverse degradation, management input is necessary to reduce the loss of essential soil resources, restore understory propagules and establish healthy, below ground components such as soil microbial communities. Many pinyon-juniper woodlands are in an advanced state of degradation where understory species have been depleted for many years and the seedbank is depauperate, resulting in a decreased likelihood of natural recovery. Unless these woodlands are artificially seeded, natural recovery may be slow resulting in an increased time span until desired future conditions are reached. Escalating seed costs and high demand for native species have compounded the problem making understory recovery difficult. Therefore, more effective means for creating ideal microsite conditions are needed to stimulate seed germination and establishment. Microsites that can provide wind barriers, aid in trapping and retaining seeds, improve soil water potential and soil nutrients will most likely increase seed germination and seedling emergence (Chambers 2000, Harper et al. 1965). Seed germination and establishment are directly correlated with the number of seeds in favorable microsite seedbeds, rather than the total number of available seeds (Harper 1977, Harper et al. 1965). By creating suitable microsites or

"islands" of elevated soil fertility for herbaceous species, land managers may contribute to the recovery of pinyon-juniper communities. The construction of fertile patches with the use of branches from woody species, which are resistant to weathering and grazing, can become a practical strategy for land managers to restore understory populations and conserve precious soil resources.

Conclusions

Results from our study indicate that slash treatments within barren, intercanopy spaces, yield less sediment loss than non-slash treatments, thus aiding in the retention of essential soil resources. Experimentally sown seeds and slash treatments together significantly increased rates of seedling establishment by altering microsite conditions, although this was only apparent for *Elymus Elymodies* and *Bouteloua curtipudula*. Changes in microsite conditions include carbon inputs and increased microbial activity. The increase in microbial communities can foster long term understory development. Seeding alone had no significant influence on the development of understory communities. Slash treatments alone also had no effects on understory development.

Other studies have also used on site slash material to recover understory populations and control sediment loss. Results have been variable. For example, Jacobs and Gatewood (1999) found a seven fold increase in herbaceous cover within the second year following the lopping and scattering of slash into interspaces. In this case, understory response to slash treatments occurred naturally, that is, seeding was not necessary in order to achieve these results. Hastings et al. (2003) demonstrated that sediment yields had significantly decreased where slash treatments had been applied. Also, four years after slash treatments were applied, a four-fold increase in herbaceous

cover was observed. In contrast, Brockway et al. (2002) reported positive effects of tree removal on understory cover but observed no significant differences between slash removal and slash dispersal treatments were observed. In an unpublished report, Huffman et al. found no significant differences in herbaceous response due to slash additions and seeding after one year. However, seeding appeared to enhance populations of perennial grasses. These variations in response to slash and seeding treatments are likely related to preexisting plant communities, viable seed banks and soil characteristics.

Implications

Results from this study clearly improved microsite conditions for the establishment of artificially sown seeds, and the conservation of soil resources. I recommend that land mangers implement treatments that will first reduce the loss of sediment within intercanopy spaces through the utilization of woody material. Slash treatments should be considered a temporary solution in aiding the recovery of pinyon-juniper woodlands. The establishment of understory vegetation is the long-term objective for recovery of these degraded ecosystems. Slash treatments can help accelerate the establishment of understory vegetation.

Although the scale at which this study was conducted is small (intercanopy patch scale), the inferences can be applied at the landscape scale. By dispersing these treatments across the landscape, one can reconstruct a diverse mosaic structure, which was historically the configuration of pinyon-juniper woodlands. However, substantial quantities of woody material could temporarily cause negative effects such as: unnaturally large fires, the promotion of bark beetle infestations and low aesthetic values. Limiting the size of treated areas and dispersing materials in optimal areas, such as the

open interspaces, could reduce these negative effects. Potentially this would alter continuous fuel loads and change fire behaviors. Cutting and allowing phloem cells to dry before or after bark beetle take flight could prevent bark beetle concerns. Eventually, prescribed fire could be introduced to eliminate excess woody fuels and mitigate aesthetic concerns. Fire should only be introduced once adequate herbaceous cover is restored.

Any of the land management recommendations from this study should be placed in the context of the objectives that are being accomplished on a site by site basis. The results from this study need to be balanced with the practically of the application in different management situations. Therefore these results do not represent the only solution but rather provide a tool that land mangers can use to promote understory vegetation and conserve soil resources. Furthermore, a variety of treatments should be considered to maintain the diversity that exists within pinyon-juniper ecosystems at the landscape level.

Future Research

Collectively, slash/mulch treatments seem to have positive effects on native understory populations and conservation of soil resources. However, ultimately it would be desirable to reach natural self-sustainability of plant communities. Whether natural biological processes eventually take over and promote "healthy" ecosystems, is yet to be seen. Can herbaceous cover reduce erosion rates to natural, sustainable levels? Do initial increases of microbial communities contribute to long term understory health and soil stability? What are the fire effects on recovered herbaceous communities that were supported by these slash treatments? What is the threshold in which slash material may support understory development? Only long term studies can effectively answer these

questions. In order to successfully restore natural ecosystem processes, the relationship between plant production and soil mechanics needs to be identified. Individual seed demographic on selective soils or slash amendments might prove to be another area of research that can help efficiently establish native understory populations. Overall, the more experiments that are implemented on the ground, the more effectively we can become at anticipating the changes as a result of these treatments.

> "Through management actions, we can slow or accelerate the trajectories of change, we can alter the direction, some times even reverse them, but we can never stop them. Every alteration we make, however will affect the type, timing, magnitude, interaction and outcome of future thresholds. The more effectively we can anticipate these changes resulting from our action, the more effective ecosystem management will be." (Robin Tausch 1999 p.363)

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